LIMNOLOGICAL STUDIES OF

DYSTROPHIC WATERS

by

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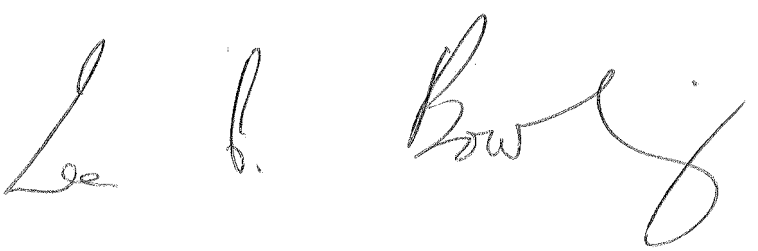
DECLARATION

This thesis contains no material which has been accepted for the award of any other higher degree or graduate diploma in any tertiary institution and that, to the best of the candidate's knowledge and belief, this thesis contains no material previously published or written by another person, except when due reference is made in the text of the thesis.



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ABSTRACT

A number of aspects of the limnology of dystrophic lentic freshwaters are covered in this thesis. Initially, studies covering a wide range of heterogeneous lakes and reservoirs from several different areas are reported, to give an overall perspective of many of the roles dissolved humic substances play in various limnological processes. More detailed case studies of a number of dystrophic lakes and reservoirs are then outlined, to demonstrate how dissolved humic substances actively influence the limnology of these individual waters.

Investigations of the underwater light climates of fifty lakes and reservoirs from Tasmania, thirty-seven from north-east New South Wales, and of twenty-six coastal dune lakes in south-east Queensland showed that dissolved humic substances were the major attenuators of light in all three areas. Increasing gilvin concentrations led to the rapid extinction of light at shallow depths, and to changes in its underwater spectral distribution from green-yellow in the clearest waters, to red in the most humic. Turbidity and phytoplankton were important contributors to attenuation in only a few of the waters studied. Results from Tasmania allowed the construction of a predictive model for use in estimating the underwater light climates of the many remote lakes of the island where *in situ* measurements are impossible, from simple laboratory measurements of small water samples. The rapid attenuation of light in humic waters also strongly affects thermal stratification, leading to strong thermal gradients, shallow thermoclines, and lengthy periods of stratification.

Chemical aspects were also examined. Strong negative correlations were found between the concentration of dissolved humic substances and pH in coastal dune lake waters from south-east Queensland, King and Flinders Islands, and western and south-west Tasmania. In contrast, there was no correlation between these two parameters in north-east New South Wales waters, probably due to lower humic concentrations and buffering by higher bicarbonate levels. Calcium and bicarbonate levels were also sometimes low in acidic, dystrophic lakes.

A wide range of photosynthetic organisms were present in the sites investigated from New South Wales, Queensland, and coastal lakes of south-west Tasmania. Phytoplankton came both from the Chlorophyceae and from other algal divisions, and desmids and dinoflagellates were

especially common. Proportions of green algae decreased relative to those with accessory photosynthetic pigments, as humic concentration increased, in the highly humic western and south-west Tasmanian lagoons, but this was not apparent in the less dystrophic waters of the other two areas. However, high humic concentrations reduced the number of phytoplankton genera present from all groups. Chrysophytes dominated the phytoplankton communities of polyhumic Lake Chisholm.

Studies of Lake Chisholm, the reservoirs of the Pieman River Power Development, and the meromictic lakes of the lower Gordon River, all in Tasmania, showed humic materials played an important role in their limnology. The rapid attenuation of light, and subsequent strong thermal stratification for much of the year lead to sluggish circulation, allowing near-meromictic conditions in Lake Chisholm, and the protection and resultant slowing of the demise of ectogenic meromixis in the Gordon River lakes. These features, along with enhancing the solubility of iron and manganese in the reducing, sulphide laden bottom waters of the Pieman River reservoirs, enabled their chemical stratification. However, other factors, such as basin morphometry and shelter, are also of importance in the limnology of these lakes and reservoirs.

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CHAPTER ONE

INTRODUCTION TO THE STUDY

1.1 AIMS AND SIGNIFICANCE

The genesis of this thesis lies in the strong influence dissolved humic substances have on the limnological face of Tasmania. The island can be divided into two distinct limnological provinces, an eastern one of predominently clear, oligotrophic waters, and a western one dominated by highly dystrophic waters. This results from a major geological, topographical, vegetational, and climatic discontinuity which occurs on a north-west to south-east axis, approximately midway across the island (Buckney and Tyler, 1973a, and references therein). Additionally, a third province, taking in all coastlines, and containing all the highly humic coastal lagoons, can also be recognized.

Dissolved humic substances bestow special properties to water, and these are the topics of this thesis. Even at low concentration they are a powerful determinant of the underwater light climate, strongly attenuating and spectrally modifying penetrating irradiance. This has permitted a classification of lakes by their optical characters predicated in large measure on a gradient of humus concentration. Highly humified lakes also display special thermal characteristics, with sharp, strong thermal gradients existing at shallow depths. This creates a predisposition towards incipient meromixis, and aids the maintenance of meromixis in several small Tasmanian lakes, and in some dystrophic reservoirs.

Additionally, the limnological effects brought about by dissolved humic substances on underwater optics, thermal regimes, and hydrogen ion concentration strongly influence the ecology of dystrophic waters, determining the flora and fauna found within them. A distinctive flora containing new genera and species occurs in the highly coloured

lakes in western Tasmania, and in the humic lagoons around the coast (Croome and Tyler, 1987a,b; Croome et al, 1987 and in press). Dystrophic and polyhumic lakes have also been identified as rich sources of phytoflagellates in Finland, and ecological and behavioural studies of these organisms have centred on them (Ilmavirta, 1980, 1983, 1984; Ilmavirta et al 1984; Jones and Arvola, 1984; Arvola 1986).

Thus, dissolved humic substances have a pervading influence on the physical, chemical, and biological character of many Tasmanian inland waters. Because of this, this thesis concentrates on dystrophy as a feature of great interest and significance in the limnology and phytoplankton floristics of the island. Additional to the investigation of Tasmanian dystrophic lakes and reservoirs, it also enquires as to whether humus exerts a similar dominant role in the physico-chemical limnology and ecology of standing freshwaters from other parts of Australia, notably those of north-eastern New South Wales and of the coastal dune areas of south-east Queensland.

1.2 THE SCOPE OF THIS THESIS

This thesis has two objectives. Firstly, it aims to give an overall perspective of the roles dissolved humic substances play in determining the character of freshwater bodies, through its influence on their physical and chemical properties, and on their biology. The second objective is to demonstrate the effect dissolved humic substances have on the limnology of some individual lakes and reservoirs. These more detailed individual studies, when considered together, also represent a progressive series which illustrate many of the features necessary to bring about the establishment and maintenance of meromixis.

Chapter Two presents reviews of the literature relevant to this thesis. However, because several different aspects of the limnology of dystrophic waters are addressed, this review takes a number of parts. Firstly, a detailed limnological introduction to the main study area, Tasmania, is provided, and the major features of lentic freshwaters from north-east New South Wales, where some comparative investigations were undertaken, are also outlined. Additionally, various limnological studies of coastal lakes of the Australian mainland are reviewed. The nature of humic substances in aqueous systems is then discussed and studies conducted on humic lakes in Finland are used to examplify the effects these have on the physio-chemical limnology and ecology of small lakes. A theoretical

background on factors affecting the spectral distribution and attenuation of underwater irradiance is also presented, and the literature dealing with the light climates of Australian and New Zealand inland waters discussed. Finally, factors contributing to the establishment of meromixis in reservoirs are examined.

The third chapter investigates factors bringing about the attenuation of solar radiation and the resultant underwater light climates of lentic inland waters. The influence of humus is examined over a series of increasingly humic lakes and reservoirs from Tasmania, north-east New South Wales, and south-east Queensland, and relationships between the light climates and attenuating factors sought. Comparisons between the three areas can also be made, as nearly all study sites in New South Wales were impoundments, while those of Queensland were natural oligotrophic coastal dune lakes. The phytoplankton of these New South Wales and Queensland waters are also reported, and their distributional trends examined in relation to the underwater light climates and nutrient levels of their habitats. The results of the survey of Tasmanian waters have been used to develop a method for predicting the light climates of remote lakes on the island where in situ measurements can not be made.

Studies into various physio-chemical and ecological aspects of humic coastal lakes and lagoons from western and south-western Tasmania, and from King and Flinders Island, in Bass Strait, are detailed in Chapter Four. Such sites represent a rich limnological resource. Optical parameters and the ionic and nutrient chemistry of these waterbodies were measured, and correlations sought between them. Comparisons of the lagoons from the islands with those of south-west Tasmania reveals similarities and differences, and the waters from both areas provide opportunities to test the empirical relationships developed between optical properties and attenuating factors in Chapter Three. The phytoplankton of some of the south-west Tasmanian lagoons was also surveyed.

Chapter Five presents more detailed accounts of investigations on individual lakes and reservoirs from western Tasmania. These include studies of polyhumic Lake Chisholm, the reservoirs of the Pieman River Power Development and Lake Barrington, and the small, dystrophic lakes along the lower Gordon River, in particular those which were meromictic. The computation of the heating and mixing

dynamics of these lakes has highlighted the dominant influence their high humic concentrations have on their physico-chemical limnology, as well as the various other factors pushing them towards meromixis, or otherwise. Some aspects of the phytoplankton ecology of Lake Chisholm are also examined briefly.

Thus, the different aspects of this study serve to illustrate the important role played by dissolved humic substances in the limnology of dystrophic waters. These are discussed in Chapter Six.

CHAPTER TWO

A REVIEW OF LITERATURE RELEVANT TO THIS STUDY

Part One

Description of Study Areas

2.1 A LIMNOLOGICAL BACKGROUND TO TASMANIA

2.1.1 The Western Limnological Province

The western area of Tasmania is a region of many low but rugged mountain ranges, with a geology composed predominantly of metamorphosed pre-Cambrian rocks. The area receives high amounts of rainfall from the prevailing westerly winds, and has cool temperate rainforest or Cyperaceous sedgelands as its dominant vegetation. Th s produce a thick mantle of peat which overlies the nutrient poor bedrock. Many of the lakes are of glacial origin, occupying cirques. Although some may vary in their chemical composition, the are for the most part dilute, with ionic proportions resembling seawater, acidic, and humified (Buckney and Tyler, 1973a,b). Most are either warm monomictic or polymictic in thermal behaviour (Tyler, 1974), but some alpine lakes may be dimictic (Steane, 1979).

Studies of four small, humic lakes amongst the cool temperate rainforest of the lower Gordon River region typify the limnological characteristics of western Tasmania. Three, Lake Fidler, Lake Morrison, and Sulphide Pool, are of special interest in that they were meromictic when first investigated (King and Tyler, 1981a, 1982a, 1983), and are the subject of further investigations reported in Chapter 5.

three lakes
These are backswamp lakes, at river level, but separated from the river by silt levee banks. Their dilute surface waters overlie more saline monimolimnions dominated by sodium and chloride ions, which originated
These ions,

from an estuarine salt wedge (King and Tyler, 1981a) which penetrated up the Gordon River during periods of low flow (Kearsley, 1978, 1982). This salt wedge raised the salinity of the riverine waters, which then entered the lakes via connecting creeks through the levee bank (Bowling, 1981; Tyler, 1986).

A notable feature of the meromixis of the three lakes was shallow chemoclines, positioned one metre or less below the surface (King and Tyler, 1981a, 1982a, 1983). The halocline spanned one metre or more of depth, but the abrupt change in redox potential between oxygenated mixolimnetic waters and reducing, sulpheretted monimolimnetic waters provided the best manifestation of chemocline position. A marked peak in bacteriochlorophyll and turbidity, due to micro-organisms was associated with the chemocline. Surface waters were humic and acid, and although concentrations of humic substances within the monimolimnion were considerably higher, pH also increased below the chemocline (King and Tyler, 1981a, 1982a, 1983).

The microstratification of chemical parameters and micro-organisms have been investigated in Lake Fidler (Baker et al, 1985a; Croome, 1984, 1986; Croome and Tyler 1984a,b; 1985a), employing thin layer samplers sampling at five centimetre depth intervals (Baker et al, 1985b). A micro-aerophylic zone of about fifteen centimetres formed the bottom of the oxycline. Dissolved sulphides appeared immediately below this, concentrations of which increased rapidly with depth. Microaerophylic and obligate anaerobic bacteria occupied distinct layers, often less than ten centimetres thick, across this oxic/anoxic interface. These included *Achromatium* cf. *oxaliferum* Schewiakoff, and cf. *Beggiotoa* sp. in the microaerophyllic zone, and *Chlorobium* cf. *limicola* Nadson and cf. *Pelodictyon* sp. immediately below the redoxcline. These organisms were responsible for the marked peak in bacteriochlorophyll (Baker et al, 1985a). Additionally, Croome (1984, 1986) and Croome and Tyler (1984a,b; 1985a) found dense populations of the euglenoid *Trachlemonas volvocina* Ehrenb., and the tiny green flagellate alga *Scourfieldia caeca* (Korsh) Belcher & Swale occupying discrete strata above the redoxcline, while a motile bacterial consortium, "*Chlorochromatium aggregatum*" Lauterborn was also a prominent member of the stratified microbial community, occurring both above and below the redoxcline. The participation of large accumulations of eukaryotic algae as permanent components of microbial stratifications is unusual (Croome, 1986). The calanoid copepod

Calamoecia tasmanica (Smith) also concentrated above the chemocline, and *Chaoborus* larvae migrated diurnally across the chemocline (Baker et al 1985a,c).

The mixolimnions of the three lakes have a considerable floristic diversity, but a generally low biomass (Croome and Tyler, 1986). Twenty-four taxa have been recorded from Lake Fidler, and sixty from Sulphide Pool, with only fifteen being common to both. Chrysophyceae were more frequent in Lake Fidler, although desmids predominated in Sulphide Pool, and together they constitute 57% of the total flora. The mixolimnions are particularly favoured habitats for silica-scaled chrysophytes, with sixteen species, or half the recorded Australian flora, being reported from them. Four occurred only in these lakes, and were frequently present in bloom proportions (Croome and Tyler, 1985b). One species, *Mallomonas tasmanica* (Croome and Tyler) Asmund & Kristiansen concentrates in a discrete zone midway in the mixolimnion of Sulphide Pool by day, but becomes more evenly distributed throughout the mixolimnion at night (Croome and Tyler, 1983).

A dam built on the mid-reaches of the Gordon River has markedly altered the flow regime in its lower sections (King and Tyler, 1982b), considerably limiting the extent the estuarine salt wedge can intrude upriver. As a consequence, the replenishment of monimolimnetic salts in the lakes is also reduced, resulting in the elimination of meromixis in Lake Morrison in early 1978. Meromixis in Lake Fidler and Sulphide Pool is likewise threatened (King and Tyler, 1982b).

The stratified micro-organisms existed at irradiance levels usually below 1% of surface incident light, and composed almost entirely of red wavelengths (Croome and Tyler, 1985a; Bowling and Tyler, 1986). Light is a limiting factor for these organisms (Baker et al, 1985a), and any reaching the depths occupied by them is rapidly absorbed (Bowling and Tyler, 1986). Decay and sinking of the chemocline has placed the micro-organisms deeper in the lakes, at depths were insufficient light is available. This factor had caused their disappearance from Sulphide Pool by 1986, while only a few of the organisms previously associated with the chemocline of Lake Fidler remained (Croome and Tyler, in press).

The fourth lake of the area, Perched Lake, is warm monomictic (King and Tyler, 1981b). It lies above river level, away from inflows of saline water. The lake is only moderately dystrophic, and although it

stratifies for almost eight months of the year, hypolimnetic oxygen is never depleted, indicating a low productivity. Its water chemistry is similar to the other three Gordon River lakes, being acidic, of low conductivity, and with the ionic proportions of seawater. The phytoplankton of Perched Lake is dominated by chrysophytes and chlorophytes, especially desmids, but no well defined seasonal changes in biomass or species composition have been observed (King and Tyler, 1981b).

2.1.2 The Eastern Limnological Province

In contrast with the western sector, the remainder of the island, comprising the Central Plateau, Midlands Graben, and Eastern Tiers, is mostly of less rugged relief, and dryer. Its geology is dominated by Jurassic dolerite, and the vegetation by schlerophyll forests or epacrid heathlands. The lakes here are much more diverse. The majority lie on the Central Plateau and are of glacial origin, or have been subjected to periglacial activity (Buckney and Tyler, 1973a). Many are large, shallow, and exposed to frequent wind action, rendering them polymictic, and even in the deepest, such as Lake St. Clair, thermal stratification never fully develops, although a few may be warm monomictic (Tyler, 1974). Most are uncoloured, with pH values close to neutral. The lakes in the highest rainfall areas are very dilute, with a seawater order of ionic dominance prevailing, but those in lower rainfall areas to the east have increased ionic concentrations, and calcium and bicarbonate as the second dominant ions (Williams, 1964; Buckney and Tyler, 1973a). A number of saline and hyposaline endorheic lagoons, with alkaline waters dominated by sodium and chloride, occur in the Midlands, where annual evaporation exceeds preceipitation (Buckney and Tyler, 1976; De Deckker and Williams, 1982).

Lakes Sorell and Crescent are two large subalpine lakes, and Lake Leake and Tooms Lake two low elevation reservoirs from the eastern limnological province, which have been the subjects of comprehensive studies (Cheng and Tyler, 1973a,b, 1976a,b; Croome and Tyler, 1972, 1973, 1975). These four lakes are all limnologically similar, being shallow, exposed, and thus polymictic, and although uncoloured are considerably turbid, causing low water transparencies. Their dilute waters show some enrichment with alkaline earth bicarbonates, but sodium and chloride are still the dominant ions, and have pH's near neutral. Lake Leake and Tooms Lake are extremely oligotrophic with sparce phytoplankton populations, and may rank amongst the world's most

unproductive lakes (Croome and Tyler, 1973, 1975). In comparison, Lake Sorell is mesotrophic and Lake Crescent eutrophic. A marked contrast exists between the planktonic floras of Lakes Sorell and Crescent, with many species restricted to either one lake or the other. Lake Crescent has ten times the biomass of Lake Sorell, composed mainly of diatoms and filamentous green algae, while desmids and chlorococcalean algae dominate in Lake Sorell (Cheng and Tyler, 1973a,b, 1976a,b).

2.1.3 The Coastal Limnological Province

The coastal areas of Tasmania form a third distinctive limnological province. The lakes within it are typically acidic, humic, and of low conductivity, but considerable temporal fluctuations may occur in these parameters. Their waters generally have ionic proportions similar to seawater, although this varies locally. The lakes of the north-east and east coasts, while still fresh, have salinities of up to 2400 mg L^{-1}, and are less dilute than the lakes of the western coastline. The East Coast lakes also display a greater influence of calcium and bicarbonate in their ionic compositions (Buckney and Tyler, 1973a; 1976; King and Civil Investigation Division, Hydro-Electric Commission, 1978b; Croome and Tyler, 1987a; Croome et al, 1987). The dystrophic, acidic coastal lake waters have been shown to be perhaps the richest phytoflagellate communities of all Tasmanian freshwaters (Croome and Tyler, 1987b). They contain a considerable number of silica-scaled chrysophytes (Croome and Tyler, 1985b), as well as other recently described genera of dinoflagellates and chrysophytes (Croome and Tyler, 1987a; Croome et al, 1987 and in press; Ling et al, in press).

The coastal limnological province also embraces the islands of Bass Strait. The geomorphological origins of the lakes and lagoons of King Island have been described by Jennings (1957). These lakes lie in dunes ranging from highly siliceous to highly calcarious in composition, and chemical analyses (Brand, 1967; Buckney and Tyler, 1973a, 1976) have shown that, while many have the ionic proportions of seawater, some lakes are enriched with calcium, bicarbonate, and sulphate. Salinites range from 200 to 1700 mg L^{-1}. Some lagoons are only slightly coloured, but others are highly dystrophic, and pH values vary from less than 5.0 to more than 9.0 in the more calcarious waters (Brand, 1967; Buckney and Tyler, 1973a, 1976). The chemical characteristics reported for lagoon waters from the islands of the Furneaux Group are almost identical (Buckney and Tyler, 1973a, 1976).

2.1.4 Additional Limnological Studies in Tasmania

Despite its many natural lakes and tarns, artificial waterbodies now make up the greater part of Tasmania's lentic waters (Kirkpatrick and Tyler, 1987). Many of these are deep, narrow and steepsided, which has contributed to the chemical stratification of some (Tyler, 1980). Most notable is Lake Barrington, where dissolved iron, manganese, alkaline earth bicarbonates, and sulphides render it meromictic (Tyler and Buckney, 1974). Morphometry related anomalous stratification behaviour may also occur, as in Lake Gordon, where cold dense underflows displace existing hypolimnetic waters upwards (Steane and Tyler, 1982). However, construction of impoundments to create new reservoirs or to raise the levels of existing waterbodies have at times been to the detriment of Tasmanian limnology, having caused the loss of the original Lake Pedder with its unique flora and fauna (Tyler, 1986), and also loss of the unusual floating island ecosystem of the Lagoon of Islands (Tyler, 1976, 1980).

Dystrophic Tasmanian lakes harbour rich collections of phytoflagellates, including new genera and species, and some of great rarity (Croome and Tyler, 1987a). Elsewhere, desmids may be important, especially in numbers of species, while the large, shallow, turbid lakes of the eastern limnological province are dominated by diatoms (Tyler, 1974). Tasmanian lakes also contain fauna of considerable interest, including Anaspididae (Syncarida), Phreatoicidea (Isopoda), and the Eustheniidae (Plecoptera) (Williams, 1964). Additionally, Galaxcidae comprise 60% of the twenty-five species of Tasmanian freshwater fish (Fulton, 1982), a number of which are endemic with very limited distributions.

2.2 A LIMNOLOGICAL BACKGROUND TO NORTH-EAST NEW SOUTH WALES

North-east New South Wales is a region containing few natural lakes, and some lagoons, but numerous reservoirs (Timms, 1970). A survey of the optical properties and phytoplankton of these waters was undertaken as part of this study, to provide comparisons with similar investigations from Tasmania, and from south-east Queensland (see Chapter 3). A brief account of the limnological features of this region is given to provide a background for this survey.

The waters of the region are dilute, with total dissolved solids usually below 300 mg L^{-1}. Sodium is the dominant cation, especially in waters near to the coast, while enrichment with calcium and

magnesium occurs in many from further inland. Bicarbonate is the dominant anion, exceeding chloride in all but a few locations close to the coast (Timms, 1970). The waters from the New England area are strongly influenced by the igneous geology of their catchments. Waters from basaltic catchments have the highest conductivities, with elevated proportions of alkaline earth bicarbonates, and pH values above neutral; while those from granitic catchments are sodium and bicarbonate dominated, with low conductivites and pH (Banens, in press). The limnological characteristics of coastal dune lakes from north-east New South Wales are given in Section 2.3, below.

The waters of the region display a wide range of clarities. Mean vertical attenuation coefficients for downwelling Photosynthetically Available Radiation (P.A.R.) (see Section 2.5.3(b), below) measured in a number of reservoirs from the upper Hunter Valley, Western Slopes, New England Tablelands, and far North Coast by Scribner (quoted in Kirk, 1986) ranged from 0.36 m^{-1} to 2.24 m^{-1}. However, corresponding colour, turbidity, and chlorophyll a data were not included. Secchi depth measurements (Scribner, in Kirk, 1986; Timms, 1970) indicate wide seasonal fluctuations occur in the optical parameters of the waters from this region of New South Wales.

Few local algal floras have been described for Australia (Ling and Tyler, 1986), and north-east New South Wales is no exception. Some observations were made, on samples sent to them, by European phycologists last century (e.g. Borge, Nordstedt, Raciborski - see Ling and Tyler, 1986), before Playfair (1914; 1915a,b; 1916a,b; 1917, 1919, 1921, 1923) compiled extensive lists and described many new species of phytoplankton from the Lismore district. These studies also included chytrids and other aquatic fungi, and some zooplankton. More recently, Skinner (1975) studied the Zygnematales (Chlorophyceae) of the New England Tablelands, and some of the diatoms have been described by Foged (1978). Ecological studies of the algae of Chaffey Dam, near Tamworth, have also been undertaken (May and Powell, 1986).

2.3 A LIMNOLOGICAL BACKGROUND TO STUDIES ON COASTAL LAKES

2.3.1 The Nature of Australian Coastal Dune Lakes

Dunes, resulting largely from Quaternary geological events, cover extensive regions of the Australian coastal lowlands (Coaldrake, 1961). Many of these areas (Figure 2.1) contain lakes, which

Figure 2.1 The location of coastal lowlands and other analogous regions in Australia, where lakes (both freshwater and saline) occur. After Coaldrake (1961).

typically
are dystrophic, acidic, of low salinity, and with ionic proportions close to seawater (Timms 1982, 1986a). Studies of acid dune lakes and swamps along the entire east Australian coastal strip have shown an assemblage of organisms from very diverse taxonomic groups which are virtually restricted to these habitats (Arthington, 1977; Timms, 1986a), while those of Tasmania have been identified (Kirkpatrick and Tyler, 1987) as being of considerable limnological significance, with specialized phytoflagellate communities (Croome and Tyler, 1987b).

Timms (1982, 1986a) classified dune lakes from the eastern Australian mainland into six types, depending on mode of origin. These include:-

1) Perched dune lakes, in elevated siliceous leached dunes where accumulated organic materials form an impervious basin floor. These lakes appear to be almost exclusively Australian;
2) Lowland dune lakes, in swales and gutters between dunes close to sealevel;
3) Watertable windows, where an interdune space dips below the local watertable, and is consequently drowned;
4) Dune-contact lakes, which overlie a solid substrate, but with dunes forming at least one shoreline;
5) Marine-contact lakes, similar to lowland dune lakes in formation, but with a present or recent connection to the sea; and
6) Frontal dune ponds, where wind created hollows in frontal dunes extend below the watertable. These ponds are often ephemeral.

However, this classification is not all-inclusive, and lakes of different origin may occur in other coastal regions (Timms, 1982, 1986a). Such lakes include dune barrage lakes, where drainage is blocked by a dune, and those occupying the axial hollows of parabolic dunes, such as on King Island (Jennings, 1957). Other lakes are of composite formation, and do not fit easily into Timms (1982, 1986a) classification.

2.3.2 Studies of Coastal Lakes of the Australian Mainland

Although the dystrophic coastal lakes of Tasmania are emerging as a very significant third limnological province (Croome and Tyler, 1987a,b; Croome et al, in press ; Ling et al, in press), information

on them is still limited, and has been reviewed in Section 2.1.3. In comparison, the coastal lakes of some parts of the Australian mainland, especially those of south-east Queensland and north-east New South Wales, have been more comprehensively studied, and therefore best exemplify the limnology of these waterbodies. Studies of the optical properties, nutrient status, and phytoplankton of freshwater coastal dune lakes from south-east Queensland and north-east New South Wales are reported in Chapter 3, and studies of Tasmanian dune lakes are the subject of Chapter 4.

While a few dune lakes occur on Cape York Peninsular, in far north Queensland (Bayly and Williams, 1972; Timms 1986b), many more are located in the siliceous coastal areas in the south-east of that state. North Stradbroke and Moreton Islands, and the Cooloola Sand Mass, each have some, but the most numerous and best examples are located on Fraser Island, which has more sizeable lakes per unit area than anywhere else in Queensland (Bayly, 1966). This sandy coastal country continues southwards along the New South Wales north coast, and several lakes are also present there.

Many of the Queensland lakes are of the perched dune lake type, but a few, however, represent watertable windows (Bayly, 1964; Bensink and Burton, 1975; Lee-Manwar et al, 1980). Lake Wabby, on Fraser Island, is a watertable window type lake, but also shows some characteristics of a dune barrage lake (Arthington et al, 1986). The New South Wales lakes are of more diverse formation, and include lowland dune lakes, dune-contact lakes, marine-contact lakes, and frontal dune ponds (Timms, 1982). Many of these lakes are being degraded by human activity (Timms, 1977a).

Water colour varies from clear and highly transparent in some lakes, especially the frontal dune ponds, to humic and strongly attenuating in perched dune lakes (Timms, 1982; Bayly, 1964; Bayly et al, 1975). Water transparencies, measured by Secchi disc, range from 0.35 to 8.20 metres (Timms, 1969; Bayly et al, 1975; Bensink and Burton 1975; Miller et al, 1976, 1984; Torgersen and Longmore, 1984; Arthington et al, 1986), but considerable temporal fluctuations occur, attributable to changes in organic staining, and to variable turbidities due to changing phytoplankton biomass (Miller et al, 1984). However, other than Secchi disc measurements, the optics of these lakes have been little studied, the exceptions being Blue and Brown Lakes, on North Stradbroke Island. In transparent Blue

Lake, the euphotic depth was approximately 10.10 metres, with green light being the least attenuated wavelengths, while the euphotic depth of Brown Lake was only 1.40 metres, and red light penetrated to the greatest depth (Bensink and Burton, 1975).

Most of the lakes are polymictic, without thermal, oxygen, or chemical stratification (Timms, 1969; Bayly et al, 1975; Miller et al, 1976, 1984). Weak thermal gradients, coupled with hypolimnetic oxygen depletion, may develop in some lakes, but these are fleeting episodes (Bensink and Burton, 1975; Arthington et al, 1986). Although some degree of thermal stability may be achieved due to their relative depths and surface area, exposure to wind and storms can cause holomixis at any time of year (Arthington et al, 1986). However, deep, well sheltered, and dystrophic Hidden Lake, on Fraser Island, is thermally and chemically stratified for most of the year, with anoxic hypolimnetic conditions (Longmore et al, 1983; Torgerson and Longmore, 1984), while Brown Lake, on North Stradbroke Island, also develops prolonged strong thermal stratification, and is probably warm monomictic (Bensink and Burton, 1975).

The freshwater coastal lakes of Queensland constitute a very homogenous group with respect to water chemistry. All are dilute, with an average salinity of about 40 mg L^{-1}, with sodium and chloride contributing almost 80% of the dissolved ions, although proximity to the coast and altitude may cause variations (Bayly, 1964; Bayly et al, 1975; Miller et al, 1984; Little and Roberts, 1983; Reeve et al, 1985; Bensink 1976; Bensink and Burton 1975; Arthington and Watson, 1982; Timms 1986b; Bayly and Williams, 1972). The atmospheric supply of ions from the sea is the major influence on lakewater chemistry, especially as the lakes lie on deep, siliceous sands, so that the supply of ions by weathering processes is negligable (Bayly, 1964; Bayly et al, 1975; Little and Roberts, 1983; Reeve et al, 1985). The silica content of Fraser Island lakewaters is low, suggesting that their waters are derived mainly from local rainfall rather than from groundwater sources. Instead, the lakes probably drain slowly but continuously into the main underground water body, thus maintaining their low salinities (Little and Roberts, 1983).

The coastal dune lakes of north-east New South Wales are more variable in their physico-chemical characteristics, due to their different types of formation. Some are humic, acidic, and of low salinity, with ionic compositions strongly dominated by sodium and

chloride. Others have higher conductivities, are less coloured and acidic, and have calcium and bicarbonate as their major ions (Bayly, 1964; Timms 1969, 1982).

The pH values of Queensland coastal lakes are generally below 6.0 (Bayly, 1964; Bayly et al, 1975; Bensink and Burton, 1975; Bensink, 1967; Miller et al, 1976, 1984; Arthington and Watson, 1982; Little and Roberts, 1983; Reeve et al, 1985; Timms 1986b; Bayly and Williams, 1972). Coloured organic acids are the principal determinants of pH. Lakes with a pH greater than 5.0 are usually colourless, while those of less than 5.0 are humic, with pH decreasing as humus concentration increases (Bayly, 1964; Bayly et al, 1975). The pH values of New South Wales coastal lakes vary considerably, but the larger ones tend mainly to be acidic (Timms, 1982).

The sparce nutrient data available indicates most of these coastal lakes to be oligotrophic, with total phosphorus and total nitrogen levels of less than 10 $\mu g\ L^{-1}$, and 1000 $\mu g\ L^{-1}$, respectively (Miller et al, 1976, 1984; Reeve et al, 1985). Some eutrophication may occur in lakes affected by recreational useage (Miller et al, 1984).

A number of biological studies have been undertaken on Queensland and New South Wales coastal lakes. Chlorophyll a contents reported for a few locations are generally low, at less than 9 $\mu g\ L^{-1}$ (Miller et al, 1976, 1984). Desmids are the most common phytoplankton present, and other green algae, dinoflagellates, diatoms, cyanobacteria, and chrysophytes have also been noted (Bayly, 1964; Bayly et al, 1975; Arthington et al, 1986). The most important component of their zooplankton is usually the copepod *Calomecia tasmanica*, which has its highest population densities in the more humic lakes. Other planktonic animals are less common, but the lakes have a richer and more diversified littoral invertebrate fauna, and fish are also present. Emergent plants are also common around the shorelines (Timms, 1969, 1982; Bayly, 1964; Bayly et al, 1975, Bensink and Burton, 1975; Bensink, 1976; Arthington, 1977; Arthington et al, 1986).

Freshwater coastal lakes located in the Gippsland and Portland regions of Victoria are also of diverse origins (Timms 1973, 1977b; Brand, 1967). This hetrogeneity is clearly evident in their chemical features. While many in Gippsland show some dystrophy, are slightly acid, and dominated by sodium and chloride, a few are

alkaline and enriched with alkaline earth bicarbonates (Timms, 1973; Brand, 1967). Those of the Portland region are also alkaline, due to enrichment with calcium and bicarbonate, and have clear, transparent waters (Timms, 1977b).

Coastal lakes in South Australia differ from those of the eastern Australian coastline in that many are markedly saline. Although fresh water lakes occur in the Beachport-Robe area (Brock, pers. comm.), others have salinities ranging from about one and a half to ten times seawater. Sodium and chloride are the dominant ions, and pH values are above 7.0 (Bayly and Williams, 1966; Bayly, 1970). Similar saline coastal lakes are also located on the Yorke Peninsular (Williams and Buckney, 1976; Tominaga et al, 1987).

A number of alkaline, saline lakes with similar ionic chemistry occur near Esperance, and along the Perth lowlands of Western Australia (Williams and Buckney, 1976). Several ephemeral and permanent lakes with seawater chemical characteristics are located on Rottnest Island (Edward, 1983), and although their salinities range from less than 1 gm L^{-1} to greater than 100 gm L^{-1}, only one is consistantly fresh. Some display a seasonal meromixis, when fresh waters from winter rains overflow their otherwise hypersaline waters (Bunn and Edward, 1984). However, there are a number of freshwater coastal lakes in south-west Western Australia. Lakes between Albany and Cape Leeuwin are considerably humic (personal observation), with conductivities between 200 and 3000 $\mu S\ cm^{-1}$ (Croome, pers. comm.), but other details of their chemistry are unknown. Lakes in the Perth area experience large seasonal changes in volume, and therefore their salinities fluctuate considerably. These lakes are alkaline, with sodium and chloride their dominant ions, although some enrichment with calcium and bicarbonate takes place, and many may be eutrophic (Williams and Buckney, 1976; Congdon and McComb, 1976; Gordon et al, 1981; Newman and Hart, 1984; Lane, pers. comm.). Although the data is scant, it appears that freshwater coastal lakes in Western Australia may differ considerably from their eastern Australian counterparts.

Part Two

A Theoretical Background to the Studies

2.4 THE NATURE OF HUMIC SUBSTANCES IN AQUATIC SYSTEMS

2.4.1 The Chemistry, Origin, and Distribution of Dissolved Humic Substances

Humic substances in lake waters originate either from autochthonous sources (formed within the lake itself), or from allochthonous sources (leached into the water from terrestrial sources) (Jackson, 1975). A complex mixture of organic substances, they may be classified into three main fractions;-

1) Fulvic acids, which are water soluable at all pH values, and have molecular weights of less than 10,000 Daltons;
2) Humic acids, which are soluable in alkaline media but precipitate at low pH, and have molecular weights ranging from 10,000 to 300,000 Daltons; and
3) Humins. These are insoluable at all pH values, have the highest molecular weights, and are present in colloidal rather than dissolved form. (MacCarthy et al, 1985; Giesy and Briese, 1978; Lawrence, 1980; Allen, 1976; Pennanen, 1982).

Elemental analysis has shown humic substances to be about 50% carbon, along with, in descending order of percent composition, oxygen, hydrogen, nitrogen, and sulphur (Schnitzer, 1978). Fulvic acids contain more oxygen and sulphur than humic acids, and less carbon. Structurally, humic substances are generally accepted to be predominantly aromatic and phenolic in character, being polymers of various carboxylic acids, benzenepolycarboxylic acids, and phenolic acids (Schnitzer, 1978).

The physical and chemical properties of dissolved humic substances are dependant on the pH and ionic concentration of the aquatic medium. Increasing ionic concentration will cause a decrease in the molecular size and weight of dissolved humic substances (Aho and Lehto, 1984; De Haan et al, 1987), while increasing pH has been shown to cause an increase in the molecular sizes and weights of dissolved fulvic acids (Ghassemi and Christman, 1968; De Haan et al, 1983). Molecular size and weight may also be altered by freezing (Giesy and Briese, 1978).

Allochthonous humic materials are derived principally from chemical and biological degredation of plant tissues, especially lignins, and from the synthetic activities of micro-organisms, with the end products often tending to be more stable than the starting materials (Schnitzer, 1978). Similar products have been isolated from both leaf litter and lake water, indicating a terrestrial origin of the dissolved aquatic humus (Hall and Lee, 1974). This humus is washed into waterbodies by lateral transport from the upper soil zone. The B soil horizon removes dissolved humus, so groundwater does not contribute to the humus content of lakewater (Cronan and Aiken, 1985; Reeve and Fergus, 1982).

Autochthonous humic substances originate from the chemical degredation of the cellular constituents and exudates (principally carbohydrates, amino acids, and lipids) of indigenous planktonic organisms and aquatic plants. The majority are water soluable fulvic acids, which differ from allochthonous humic substances by having higher nitrogen contents (Kalle, 1966; Jackson, 1975; Ward and Wetzel, 1984). Autochthonous humic substances may constitute a large proportion of the dissolved yellow substances of eutrophic lakes with high phytoplankton productivity (Davies-Colley and Vant, 1987).

The structural chemistry of dissolved humic substances may vary considerably geographically, even within small localized areas (Szpakowska et al, 1986). Low molecular weight compounds predominate in most lakewaters (Lawrence, 1980; Hama and Handa, 1980; Pennanen, 1982; Allen, 1976; De Haan and De Boer, 1986), but the proportion of higher molecular weight compounds rises as the humus concentration increases, and these may comprise a large percentage of the humic substances present in polyhumic lakes (Pennanen, 1975; Aho, 1986; De Haan et al, 1987). In some instances, colloids may account for over 50% of the humic substances present (Koenings and Hooper, 1976; Lock et al, 1977). The proportion of high molecular weight compounds also increases with depth (Pennanen, 1975, 1982; Aho, 1986), and seasonal fluctuations, both in total humus content, and in the relative proportions of humic to fulvic acids, can be marked (De Haan, 1972; De Haan and De Boer, 1979, 1986; Pennanen, 1982).

2.4.2 Humic Substances and Colour

Humic substances dissolved in water have a yellow to brownish colour due to the oxidation of various phenolic constituents, principally hydroxyhydroquinones, to form deeply coloured intermolecular

quinhydrones (Flaig, 1975). Mixtures of substances thus formed show similar, nearly monotonic spectra with absorbance increasing with decreasing wavelengths of blue and ultra-violet radiation. These spectra are approximately exponential in shape, and remarkably similar regardless of origin (Davies-Colley and Vant, 1987), and coloured humic substances are in large measure quantitatively indistinguishable over a wide range of latitudes (Lewis and Canfield, 1977).

The larger molecular weight humic compounds are generally the most highly coloured (Hall and Lee, 1974; Ghassemi and Christman, 1968; Pennanen, 1975; Aho, 1986). Humic acids have a greater intensity of colour than fulvic acids, and colloidal humus may also contribute greatly to water colour (Koenings and Hooper, 1976; Lock et al, 1977), although low molecular weight fractions also significantly colour waters where they are the predominant humic substances (Hama and Handa, 1980; Wheeler, 1976). However, the colour of humic substances in water varies with pH, increasing as pH increases, and decreasing with increased acidity (Bayly, 1964; Ghassemi and Christman, 1968; Hall and Lee, 1974). Chelation with iron also increases the colour of dissolved humic substances (Hall and Lee, 1974).

2.4.3 Humic Substances and pH

The presence of dissolved humic and fulvic acids can make a significant contribution to the acidity of water, adding markedly to the hydrogen ion concentration down to a pH value of about 4.0 (Glover and Webb, 1979). The main functional groups of humic substances which increase the acidity of natural waters are their carboxylic acids and phenolic hydroxyls. The extent to which these dissociate varies with pH. Greater ionization occurs with decreased pH, due to the wide range of similar, but non-identical functional groups present in aquatic humic materials (Oliver et al, 1983; Perdue et al, 1980).

2.4.4 Humic Substances and Metal Ions

The solubility of various metal carbonates and sulphides, especially those of iron and manganese, may be greatly enhanced by the presence of humic substances, and the solubility of their hydroxides is also improved slightly (Rashid and Leonard, 1973). The various acidic functional groups of humus molecules are considered responsible for this. Low molecular weight molecules

have a greater metal-holding capacity than larger fractions, but this also depends on pH, redox potential, and on the metal itself. Divalent metals are complexed considerably better than trivalent metals (Rashid, 1971; Jackson, 1975; Szpakowska et al, 1986). However, complexation may not always account for the greater solubility of the metal ions, which can be present in a dissolved state unassociated with humic substances (Koenings and Hooper, 1976; De Haan and De Boer, 1986). Dissolved humic substances may reduce some metal ions, or prevent the oxidation of some reduced species (Szilágyi, 1971; Koenings and Hooper, 1976).

In comparison, alkaline earth metals, even at low concentrations, and sodium, actively coagulate and precipitate dissolved humic substances (Sholkovitz and Copland, 1981), and humus can be removed from lakewaters by precipitation with calcium carbonate and ferric oxides (Otsuki and Wetzel, 1973; Tipping, 1985).

2.4.5 The Ecological Significance of Humic Substances

The presence of dissolved humic substances in lakewaters may have considerable ecological significance. Specialized floras and faunas are found in humic lakes in North America (Patrick et al, 1981), Australia (Timms, 1986a, Croome and Tyler, 1987b) and in polyhumic forest lakes in Finland (Ilmavirta, 1980, 1983, 1984; Ilmavirta et al, 1984; Jones and Arvola, 1984; Arvola, 1986). A number of frequently conflicting hypotheses have been presented to explain the ecological influences of dissolved humic substances.

Several adverse effects have been claimed, especially on primary productivity, which typically is low in humic waters (Prakash et al, 1975; Salonen, 1984). Dissolved humic substances strongly absorb blue light, decreasing that available to chloroplast pigments of phytosynthetic aquatic organisms which utilize these same wavelengths. The ability of some aquatic primary producers to photosynthesize and grow may thus become progressively more impaired as the concentration of humic substances within the water increases (Kirk, 1976b). However, the remaining light climate, dominated by red wavelengths, may be exploited by those with the necessary accessory photosynthetic pigments to do so, and it has been speculated (Eloranta, 1978; Kirk, 1977b, 1979, 1981c; Jeffrey, 1980) that the characteristic pigment arrays of different classes of algae place each of them at an advantage somewhere in the range of light climates encountered in natural waters.

Additional detrimental effects suggested include the immobilization of micronutrients by their formation of strong, stable complexes with humus (an excess of a chelator reduces the availability of trace elements to algae); excessive acidity of the water; or the presence of antibiotic substances such as phenols (Prakash et al, 1975; Jackson, 1975). All these may inhibit primary productivity.

On the other hand, beneficial effects ascribed to the presence of humic matter in aquatic systems are often contradictory to the adverse effects proposed. This includes the suggestion that humic substances may make metal trace element nutrients more available to algae, by enhancing their solubility (Jackson, 1975; Prakash et al, 1975). Metal-humate complexes may also influence the availability of phosphorus, either through co-adsorption, or by allowing orthophosphate to remain free in solution (Prakash et al, 1975; Koenings and Hooper, 1976; De Haan and De Boer, 1986). Complex formation and removal of heavy metals such as cadmium and lead may also be ecologically important (Szpakowska et al, 1986), and it has been shown (Gjessing, 1981) that algal and fish toxicity to cadmium is considerably reduced in the presence of dissolved humus.

While large concentrations of dissolved humic substances inhibit algal productivity (Prakash et al, 1975; Salonen, 1984 - see above), small concentrations, especially of low molecular weight humic acids, and to a lesser extent fulvic acids, produce positive growth responses. These include extended exponential growth phases and increased growth rates in planktonic marine diatoms and dinoflagellates, leading to greater cell yields, when grown in the presence of small amounts of humic substances. The dissolved humic fractions may act as stimulants for algal cells and be involved in cellular metabolic processes, or could supply micronutrients which might otherwise be unavailable (Prakash and Rashid, 1968; Prakash et al, 1973, 1975).

Humic compounds may provide an energy source for some aquatic organisms, especially filter and detrital feeding forms, particularly if other organic molecules such as proteins and polysaccharides are associated with them (Prakash et al, 1975). Humic compounds may form the basis of significant food webs, utilized by hetrotrophic bacteria, which are in turn fed upon by heterflagellates and zooplankton, with even phytoflagellates supplementing their autotrophy with phagotrophy (Bayly, 1964; Arvola, 1985). The extent this occurs depends on the nature and content of the humic substances, which are usually considered

refractory and biologically non-utilizable. Generally only 20 to 30% of dissolved humic compounds are rapidly biodegradable in natural, unpolluted waters, but may be much lower in some waters (Servais et al, 1987). However, even if only a small fraction of the vast store was biologically available, it would still comprise a significant contribution to the food webs of humic lakes (Salonen, 1984).

2.4.6 Limnological Features of Humic Lakes - Examples from Finland

Finland has many lakes, most of which are small and highly humic due to the input of water from surrounding forests and peatlands (Ilmavirta 1979, 1982). As such, they provide an appropriate introduction for studies of humic lakes from other areas such as Tasmania, and serve to highlight many of the limnological features displayed by small polyhumic forest lakes.

The brown colouration of Finnish lake waters markedly influences the penetration of light within them, with the vertical attenuation coefficient for Photosynthetically Available Radiation (P.A.R.) being closely linearly related to water colour (Eloranta, 1978; Jones and Arvola, 1984). The proportion of longer wavelength light increases as colour increases, while the depth of the euphotic zone (the depth through which 99% of incident illumination is attenuated) decreases very rapidly at first with initial colour increases, until a certain point, when additional increases have little further effect.

The pattern of light penetration has important consequences for the mixing regimes of these small, sheltered, polyhumic forest lakes (Jones and Arvola, 1984; Salonen, Arvola and Rask, 1984). In the absence of wind induced circulation, an inverse relationship occurs between water colour and depth of mixing, so that the euphotic zone often equals the epilimnion in volume, due to the dependance of mixing depth on that of light penetration. The rapid attenuation of solar radiation close to the surface produces steep thermal gradients, which may form very early in spring, immediately after the break-up of winter ice. This reduction of the vernal circulation period often leads to incomplete mixing and a tendency towards "spring meromixis" in what would otherwise be dimictic lakes (Salonen, Arvola, and Rask, 1984; Salonen, 1984).

Because of the strong thermal stratification, and possible "spring meromixis", the hypolimnetic waters, which can comprise much of the volume of these lakes often become anoxic (Salonen, Arvola, and Rask, 1984;

Arvola, 1985). Consequent to this, a highly stratified nutrient chemistry occurs. Concentrations of phosphate, ammonia, and nitrate-nitrite are usually very low in the oxic, photosynthetically active surface waters, but abundant free nutrients exist in the dark, anoxic hypolimnion (Ilmavirta, 1983, 1984; Salonen, 1984; Arvola, 1985). Convective mixing within the epilimnion at night may bring some nutrients up from deeper waters to replenish the surface waters (Arvola et al, 1987).

A further important feature of the water chemistry of Finnish polyhumic forest lakes is their generally low pH, with the most highly coloured being the most acid (Ilmavirta, 1980, 1983). While the mean pH value is around 6.0 (Ilmavirta, 1980, 1983; Ilmavirta et al, 1984; Arvola, 1985, 1986), that of individual polyhumic lakes can be as low as 4.4 in some instances (Arvola, 1983, 1984a,b; Rask et al, 1986). However, pH tends to increase with depth in such lakes, often by as much as 1.0 unit.

Finnish polyhumic forest lakes are generally dilute, with conductivities of less than 50 $\mu S\ cm^{-1}$ (Ilmavirta, 1980; Ilmavirta et al, 1984; Arvola, 1986), although these may increase with depth, resulting either from biogenic meromixis, or from a buildup of solutes (e.g. Fe^{2+}) during summer stratification and hypolimnetic anoxia (Arvola, 1983, 1984a; Rask et al, 1986). Calcium is the dominant cation, and sulphate and bicarbonate the dominant anions (Ilmavirta et al, 1984; Rask et al, 1986).

Ecological studies of phytoplankton composition and productivity (Ilmavirta, 1980, 1983, 1984; Arvola et al, 1987) show flagellates to be the most important contributors to both biomass and species numbers in many dark and oligotrophic Finnish lakes, with the percentage of these in the algal biomass increasing as water colour increases. Flagellates usually comprise more than 50% of algal biomass once water colour exceeds 50 $mg\ L^{-1}$ Pt., and are very common in small lakes where low turbulence enables them to exercise control over their position in the water column (Jones and Arvola, 1984; Arvola, 1986). Flagellates may also have an ecological advantage in acidic waters (Ilmavirta, 1984). Only a few species dominate the whole community, the most important being from the Cryptophyceae and Chrysophyceae (Ilmavirta et al, 1984; Ilmavirta, 1984; Arvola et al, 1987), although flagellated chlorophytes also comprise a significant proportion of the algal biomass following ice-melt (Arvola, 1986).

Rapid seasonal changes in the physico-chemical regimes of Finnish polyhumic forest lakes make them unfavourable for any particular phytoplankton community, and therefore continuous changes in species composition take place (Jones and Ilmavirta, 1978a; Ilmavirta, 1982). This succession, described for a number of Finnish lakes (Arvola, 1983, 1984a; Arvola and Rask, 1984; Ilmavirta, 1982; Jones and Ilmavirta, 1978b; Rask et al, 1986) shows general trends. Initial blooms occur soon after ice-melt, utilizing the sudden increase in light and temperatures, but are succeeded by other species better able to cope with the declining epilimnetic nutrient levels as summer progresses. Further changes occur as solar radiation and temperatures decrease in autumn. Flagellates are always dominant during the most difficult and critical periods of the year, such as at ice-melt and during the last days of stagnation (Ilmavirta, 1984), but may be so at any other time, too (Arvola, 1986). The structure of the phytoplankton community may also be affected by the rapid throughflow of epilimnetic waters (Arvola, 1985).

Daily vertical migrations of phytoflagellates also occur in humic Finnish lakes. Cryptomonads in particular move downwards in the afternoon and up again in the early morning, traversing a depth of two metres and a 15°C temperature gradient, responding to changes in light intensity. In comparison, the diurnal vertical distributions of non-motile algae remain unchanged (Arvola, 1984b; Salonen, Jones and Arvola, 1984; Arvola et al, 1987). Ilmavirta (1983, 1984) suggested that vertical migration enables flagellates to find optimal light conditions, but also hypothesized the possibility of nutrient retrieval from hypolimnetic waters. Cryptomonads have been shown to transport radio-active labelled phosphorus from depth to the surface (Salonen, Jones and Arvola, 1984).

The scarcity of epilimnetic nutrients, and the shallow euphotic zone, severely retard phytoplankton productivity in these lakes. Productivity is greatest close to the surface, and decreases rapidly with depth (Arvola, 1983, 1984a,b; Ilmavirta, 1982). Diel variations occur due to changing light intensities during the day, and also because migrating phytoplankton shift their production to different depths (Ilmavirta, 1982, Arvola, 1984b, Arvola et al, 1987). Variations in climatic and biological parameters (e.g. species composition) produce marked daily fluctuations in primary productivity

(Ilmavirta, 1978), while seasonally production follows changes in solar radiation and water temperature, increasing from a spring low to a summer maximum before declining again in autumn (Ilmavirta, 1982; Arvola, 1984a). However, several peaks and dips in production may occur throughout the spring and summer seasons (Arvola, 1983). While radiant energy determines the seasonal pattern of productivity, the nutrient concentration roughly determines the level (Ilmavirta, 1979, 1982). One source of nutrients may come from the bacterial decomposition of organic allochthonous inputs (Ilmavirta, 1979, 1981, 1982; Salonen, 1981). Additionally, the rapid decomposition of autochthonously produced organic materials within the epilimnion of these lakes means rapid recycling of nutrients takes place (Ilmavirta, 1981, 1983; Salonen, 1981; Arvola, 1984a).

The input of energy by allochthonous materials into polyhumic lake ecosystems is considerable, and may exceed that gained by net phytoplankton production by several times (Sarvala et al, 1981). Plankton respiration in such lakes is also considerably greater than phytoplankton productivity (Salonen et al, 1983; Salonen, 1984), made possible by the utilization of allochthonous sources of organic carbon by bacterioplankton (Salonen, 1981).

The low light levels and reduced turbulence of polyhumic lakes may produce physiological effects on phytoplankton. Those in the near stagnant waters close to the bottom of the euphotic zone have intense pigmentation, ensuring efficient utilization of the little available radiation (Jones and Ilmavirta, 1978b). Phytoplankton also have an increased photosynthetic efficiency and are more productive relative to the available light and their chlorophyll a content at depth than when closer to the surface (Arvola, 1984a,b; Arvola et al, 1987). However, production efficiency varies considerably diurnally due to complex interactions between light intensity, nutrient concentration, and vertical migrations (Arvola, 1984b). In comparison, adaptations by phytoplankton for photosynthesis at particular irradiance levels do not occur in lakes with considerable circulation of the water column (Jones and Ilmavirta, 1978b). The efficiency to utilize irradiance also varies annually, but due to changes in species composition rather than to changing irradiance (Jones and Ilmavirta, 1978b).

2.5 FACTORS DETERMINING THE UNDERWATER LIGHT CLIMATES OF LAKES

2.5.1 The Fate of Light in Water

"Light" refers to that portion of the solar spectrum to which the human eye is sensitive, comprising wavelengths between about 400 and 700 nm. These wavelengths also correspond approximately to those utilized by plants for photosynthesis and termed "Photosynthetically Available Radiation" (P.A.R.) (Kirk, 1983).

Light penetrating water becomes diminished in intensity until extinguished. Two factors, absorption and scattering, contribute to this attenuation. Absorption is the process whereby a photon is "captured" by a molecule of water or of some substance dissolved or suspended in it. The light energy exists briefly as electronic excitation energy within the molecule before either being released as heat, or converted to chemical energy by photosynthesis. A tiny portion may be converted to light again by fluoresence, but this is usually re-absorbed (Kirk, 1983).

Scattering is when a photon interacts with some component of the medium so that it diverges from its original path without being absorbed. Rather than removing light, scattering impedes its vertical penetration, causing photons to proceed in a zig-zag path as they bounce from one scattering particle to the next. This increases the total pathlength a photon must travel to traverse a given depth, thereby increasing its chance of absorption en-route. Some photons may also be scattered back, in an upwards direction and out into the air (Kirk, 1983).

2.5.2 Components of the Aquatic Medium Causing Absorption and Scattering

Four components of the aquatic medium attenuate light. These are:-

1) The water itself. Pure water is a weak attenuator of blue and green light, but its absorption increases at wavelengths above 550 nm, and becomes quite significant in the red region of the visible spectrum (Kirk, 1983). Molecules of pure water also scatter light very weakly, but this effect is not significant compared to its absorption. Scattering decreases with increasing wavelength (Smith and Baker, 1981).

2) Dissolved yellow substances, termed "gilvin" by Kirk (1976b), strongly absorb wavelengths below 500 nm, and longer wavelengths are also removed as concentration of gilvin increases (Kirk, 1976b).

3) Inanimate particulate matter (tripton) contributes both to absorption and scattering. Inanimate suspensoids have absorption spectra similar to gilvin, strongly attenuating the shorter wavelengths due to insoluable humus materials adsorbed onto the surface of suspended mineral particles (Kirk, 1980a, 1985; Davies-Colley, 1983). Tripton is responsible for the majority of light scattering in natural waters, increasing attenuation by 20-200% above that expected from absorption alone in inland waters (Kirk, 1981a, 1983; Davies-Colley, 1983).

4) Photosynthetic biota (Phytoplankton, and macrophytes where present) both absorb and scatter light. The major photosynthetic pigments, chlorophylls, absorb mainly blue wavelengths, and also a narrow band of red light. Accessory pigments include the carotenoids, which absorb wavelengths up to 500 nm, and the biliproteins, with absorption spectra centred about 600 nm (Kirk, 1983). Different taxonomic groups of the photosynthetic biota display considerable variations in the wavelengths they can utilize, resulting from differences in pigment composition (Kirk, 1983; Davies-Colley et al, 1986).

The capability of phytoplankton to absorb light depends on several factors, including the "packaging" of their pigments within the chloroplasts of their cells; their size; and their shape (Kirk, 1975a,b, 1976a). While the absorption spectra of species from within the same taxonomic group are similar, the values of specific absorption coefficients differ considerably between species over most wavelengths (Davies-Colley et al, 1986).

Phytoplankton cells also scatter light. Different scattering patterns occur between algal groups, but similar patterns occur within groups. The scattering spectra of phytoplankton cells generally have an inverse relationship to their absorption spectra, resulting in part because absorption removes various wavelengths from the incident beam which would otherwise by available for scattering (Davies-Colley et al, 1986). Phytoplankton may contribute more to underwater light

attenuation through scattering than through absorption, leading to the rapid extinction of light at only shallow depths in eutrophic lakes (Davies-Colley *et al*, 1986).

The effects of macrophytes on the underwater light field varies greatly, depending on plant growth habit and morphology, although attenuation is considered lower per milligram of chlorophyll in macrophytes than in phytoplankton (Kirk, 1983). However, dense stands of floating and emergent macrophytes can quickly attenuate underwater light.

2.5.3 The Optical Properties of the Water Column

(a) Inherent Optical Properties

These optical properties belong to the water itself, being determined entirely by it composition, and whose values are not affected by the prevailing distribution of light (Kirk, 1982, 1983, 1984; citing Preisendorfer, 1961). The most important of these are the absorption coefficient, a, and the scattering coefficient, b.

When a thin, parallel beam of monochromatic light illuminates an infinitesimally thin layer of water, some of it is absorbed by the layer, and some of it scattered. The fraction of the incident light beam which is absorbed, divided by the thickness of the layer is the absorption coefficient, a; while the fraction which is scattered, divided by the thickness of the layer, is the scattering coefficient, b (Kirk, 1977a, 1982, 1983). The value of the absorption coefficient for a given aquatic medium may change considerably with wavelength, due to the presence of various absorbing components (e.g. gilvin or tripton (Kirk, 1976b, 1980a)), but the scattering coefficient is fairly uniform throughout the bandwidth of P.A.R. (Davies-Colley, 1983; Phillips and Kirk, 1984). The beam attenuation coefficient, c, is the sum of both the absorption and scattering coefficients ($c = a + b$) (Kirk, 1982, 1983).

A fourth inherent optical property, the volume scattering function $\beta(\theta)$, describes the angular distribution of scattering from that point in the thin layer on which the light beam in incident. It has a characteristic shape in any given medium, forming a radially symmetrical cone around the direction of the beam, specifying the probabilities of scattering at different angles (Kirk, 1981a). In all natural waters investigated, $\beta(\theta)$ is such that most scattering occurs in a forward direction, with about 50% at angles of less than 5° to the direction of the incident light (Kirk, 1982, 1983).

(b) Apparent Optical Properties

These are the optical properties of natural waters most commonly measured. However they are not actual properties of the water itself, but of the light field established within it, but their values are determined predominantly by the inherent optical properties. They vary to some extent with both depth and changes in the incident radiation field, such as solar elevation (Kirk, 1982, 1983, 1984).

The vertical attenuation coefficient, K, (Kd for downwards irradiance, Ku for upwards irradiance) is an important apparent optical property expressing the rate light is diminished with depth. It provides a useful approximation of light attenuation attributable to a waterbody itself, and as such can be used to compare one waterbody at a given time with another, or as a guide to temporal changes in the optical character of a particular waterbody (Kirk, 1986). With vertically incident light and an absence of scattering, K would equal the absorption coefficient, a.

A further parameter at times regarded as an apparent optical property is irradiance reflectance, R, a measure of the ratio of upward to downward irradiance, at any specified optical depth. Whether a waterbody looks clear or turbid is a function largely of irradiance reflectance, especially that occurring just below the surface (Kirk, 1986).

The relationship between inherent and apparent optical properties is significantly dependent on the angle of the light flux incident on the water surface (Kirk, 1981b, 1984). As the angle departs increasingly from vertical, both absorption and scattering intensifies as photons travel more obliquely and less vertically. This increases their average pathlength per metre of depth, and thus their probability of being absorbed or deflected by a particle or molecule. If absorption remains constant, and solar elevation decreases, the value of Kd increases, but becomes progressively less responsive to increased scattering. The rate of increase of Kd is dependant on the aquatic medium. In highly scattering inland waters, Kd is rather insensitive to changes in solar altitude, but in clear, oceanic waters, where scattering is low in comparison to absorption, Kd can vary significantly with the angle of the sun. Likewise, irradiance reflectance also increases with decreasing solar altitude, because as the incident beam moves away from vertical, an increased proportion of the more intense forward scattering (due to the shape of the

volume scattering function) also moves away from the vertical, following the angle of the incident beam, and becomes upwards rather than downwards scattering (Kirk, 1984).

2.5.4 The Attenuation of P.A.R. with Depth

Monochromatic light penetrating a homogenous aquatic medium is attenuated exponentially with depth, in accordance with Beer's Law, $E_{(z)} = E_{(o)}e^{-kz}$, where $E_{(z)}$ and $E_{(o)}$ are the values of irradiance at depth z and just below the surface, respectively, and K the vertical attenuation coefficient for that wavelength of light. Thus a plot of the natural logarithim of $E_{(z)}$ against depth should be linear.

In comparison, P.A.R. includes all wavelengths from 400 to 700 nm and as it penetrates a waterbody, it becomes progressively spectrally impoverished as the most rapidly attenuated wavelengths are removed close to the surface, leaving only those least attenuated remaining at depth (Kirk, 1983; Talling, 1982, 1984; Howard-Williams and Vincent, 1984). Concomitant with this, the attenuation rate of P.A.R. is greatest in the surface waters, and decreases with depth. Thus a plot of ln $E_{(z)}$ against depth is curved rather than linear, with Kd decreasing with depth. This is most readily apparent in non-turbid waters, either clear or coloured (Kirk, 1977a,b, 1983). However, it is less apparent in turbid waters where there is a countervailing tendancy for attenuation, at all wavelengths, to increase with depth as the downward flux becomes more diffuse through scattering (Kirk, 1977a,b, 1983). Additionally, due to the rapid absorption of blue light by tripton, the spectral modification of P.A.R. may occur so close to the surface that the change in attenuation with depth is not readily detected.

In most waters, even where attenuation of P.A.R. is noticeably biphasic, it varies only slightly from the exponential model predicted by Beer's Law for monochromatic light (Kirk, 1983). Attenuation of P.A.R. for a given waterbody can therefore generally be best characterized by a single, depth-averaged value of Kd (Kirk, 1977b, 1983, 1986).

2.5.5 Studies of Light Attenuation in Australian and New Zealand Inland Waters

Extensive data on underwater quantum irradiance, and measurements of its spectral distribution are few in number (Roemer and Hoagland, 1979; Jewson *et al*, 1984), and although Kirk (1986) gave an extensive list of

optical properties for Australian inland waters, few of the studies he detailed gave information on actual underwater quantum irradiance. This section reviews those which have been undertaken on Australian and New Zealand freshwaters to provide a background to the studies of the light climates of lentic freshwaters from Tasmania, north-east New South Wales, and south-east Queensland, reported in Chapter 3.

Initial studies by Kirk (1977b, 1979) examined the light climates of some inland and coastal waters from south-east New South Wales and the Australian Capital Territory. Attenuation was weak in the clear, non-turbid coastal waters, with peak transmission being about 550 nm. In comparison, attenuation was much greater in the inland waters, but changed considerably with temporal fluctuations in turbidity and gilvin concentration. For example, the depth averaged Kd ($Kd_{(av)}$) varied by eight times over a three year period in Lake Burley Griffin (Kirk, 1980b). Consequently the spectral distribution of the underwater light also changed, being centred at about 600 nm when turbidity was low, and at 700 nm when it was high (Kirk, 1979). Gilvin absorbs the greatest proportion of P.A.R. in most waters from this area (Kirk, 1980a). Large temporal variations in optical properties, including $Kd_{(av)}$, turbidity, and Secchi disc depth, also occur in waters from other areas of New South Wales (see Kirk, 1986).

A spatial concentration gradient of gilvin, chlorophyll a, and suspended solids results in changing light climates through the Gippsland Lakes, Victoria (Hickman *et al*, 1984). P.A.R. is most strongly attenuated in Lake Wellington, furtherest inland, and this lake has a predominantly yellow spectral distribution with depth. In comparison, attenuation is least in Lake Victoria, nearest the sea, and its underwater spectrum is centred on blue-green light. Despite seasonal variation, tripton is the main attenuating component in these lakes, although water itself is important in removing red light in the clearer waters of Lake Victoria.

Light extinction is extremely rapid in many inland waters of south-east South Australia, especially in the turbid Murray River and Lake Alexandrina (Ganf, 1980; Geddes, 1984). Reservoirs of the Adelaide area are clearer, although they still have euphotic depths of less than four metres, with light being extinguished by gilvin and suspended particulate matter, and blooms of phytoplankton also contribute significantly at times (Ganf, 1980). Considerable seasonal variation occurs in the optical parameters of these waters too.

Turbidity is also the dominant parameter affecting the light climates of Northern Territory billabongs, as gilvin concentrations are always low, and phytoplankton blooms infrequent. Their tripton loadings increase greatly as they contract due to evaporation during the dry season, specifying an underwater spectrum of predominantly red light and high $Kd_{(av)}$ values (Walker, 1984; Walker and Tyler, 1984). Absorption by particulates is usually greater than absorption by gilvin, and scattering by the particulate matter markedly increases $Kd_{(av)}$ for P.A.R. in most billabongs (Kirk and Tyler, 1986).

Studies from the Mt. Isa, Townsville, and Palm Island areas of Queensland (Finlayson et al, 1984; Finlayson and Gillies, 1982; Hawkins and Griffiths, 1986) report considerably lower attenuation than in the turbid waters from other parts of Australia, with tripton being a minor attenuating component. However, seasonal changes are still apparent, resulting especially from the effects of stratification of the water column, and from periodic epilimnetic blooms of phytoplankton. Absorption by phytoplankton at times contributes to over 40% of the attenuation in Solomon Dam, Palm Island, and gilvin was also present at moderate levels in this reservoir (Hawkins and Griffiths, 1986).

The underwater light climates of fifteen mainly montane Tasmanian lakes were measured by Steane (1979). Their optics vary, depending on whether they lie in either the eastern or western limnological province of the island (see Section 2.1). Those in the east are extremely clear, with very low attenuation, and with their spectral distribution of P.A.R. at depth centred between 550 and 600 nm. Attenuation was more rapid in the dystrophic lakes in the western limnological province, leaving a predominantly red underwater light field. The dystrophic lakes of the lower Gordon River area have similar underwater light climates (King, 1980; King and Tyler, 1981b, 1982a, 1983; Croome and Tyler, 1984a, 1985a), although the characteristic stratification of photosynthetic micro-organisms at the chemocline of the meromictic lakes here markedly increases the attenuation of any light penetrating to these depths (Bowling and Tyler, 1986).

Surveys of the underwater light climates of New Zealand lakes (Vant and Davies-Colley, 1984; Howard-Williams and Vincent, 1984) reveal many to have high clarity waters, where attenuation of P.A.R. is very low. Others however are turbid, due either to phytoplankton dominated seston or to tripton, and including geothermal lakes with

varying amounts of suspended sulphur. Attenuation in these is similar to that in turbid Australian lakes, with light penetrating only to shallow depths. Humic-stained forest lakes from the South Island also rapidly attenuated light. Spectral changes in P.A.R. occur, with Lake Coleridge, the clearest lake, having a predominantly blue light field at depth. Green light is the most penetrating wavelength in other clearwater lakes and red light is transmitted best in the more turbid, coloured, or eutrophic lakes (Howard-Williams and Vincent, 1985). Considerable temporal changes in attenuation, with $Kd_{(av)}$ varying by up to five times, have been demonstrated in some lakes (Vant et al, 1986). This variation was usually closely correlated to changes in the chlorophylla content of the water.

2.6 A BACKGROUND TO CHEMICAL STRATIFICATION AND MEROMIXIS

2.6.1 The Nature of Chemical Stratification

Chemical stratification of lakes and reservoirs, usually known as meromixis, is the division of the water column into separate strata created by density differences arising from different concentrations of dissolved solutes in the water of each stratum. Waters with the greatest concentrations have the greatest density, and form the bottom stratum, or monimolimnion, which is overlain by the more dilute and less dense waters of the mixolimnion. The two strata are separated by a chemical gradient known as the chemocline. The density difference between the mixolimnion and monimolimnion may be sufficient to prevent their mixing, even during winter when thermal stratification is absent and wind induced circulation is at its greatest. This results in a permanent layer of dense, stagnant, usually anoxic water at the bottom of the lake, which may be compared to an unstratified lake, where circulation would normally reach even the deepest waters.

Meromixis may be classified into two distinct types: ectogenic and endogenic (Walker and Likens, 1975). Ectogenic meromixis is a condition created by external forces acting on a lake, such as a fresh water inflow over a pre-existing saline layer, or a saline underflow into a fresh water body. This type of meromixis may also be created by subsurface inflows, either of fresh or saline water, from a spring in the lake bed (known as crenogenesis); while a third mechanism, triptogenesis, involves a density difference created by an inflow of turbid water settling at the bottom of a lake. In contrast, endogenic meromixis is produced by conditions within the lake itself, with

chemical stratification arising from the decomposition of organic materials in the deeper waters (biogenic meromixis), or by the deepwater accumulation of salts precipitated by "freezing out" from a surface layer of ice (cryogenic meromixis). Biogenic meromixis is often aided by the size and shape of the lake basin, and by local catchment topography and forests, which reduce the amount of wind induced circulation in a lake. This has been coined "morphogenic meromixis" by Northcote and Halsey (1969).

This thesis looks at the role dissolved humic substances play towards the establishment and maintenance of chemical stratification in a number of meromictic or near-meromictic Tasmanian lakes and reservoirs (Chapter 5). Of particular interest is the establishment of meromixis in impoundments.

2.6.2 Meromixis in Impoundments

Meromixis is an unusual phenomenon in water bodies, and as a result, few meromictic impoundments have been documented. Tyler and Buckney (1974) reported the rapid development of biogenic meromixis in Lake Barrington, Tasmania, soon after the formation of the reservoir in 1969. The monimolimnetic water, localized at the base of the dam, was highly reducing, showed unusually large temperature rises, and considerable enrichment of such chemical species as iron, manganese, hydrogen sulphide, calcium, and bicarbonate. The meromixis persisted for several years, but had become very weak by 1978 (Tyler, 1980). A second reservoir, Lake Rowallan, also developed anoxic bottom waters with dissolved iron and manganese soon after filling, but this was removed by extensive drawdown from low-level offtakes (Tyler and Buckney, 1974).

Biogenic meromixis also occurred soon after the filling of Klíčava and Fláje Reservoirs, Czechoslovakia. Biochemical decomposition of organic matter lead to anoxia and the concentration of dissolved solids, especially manganese, in their bottom waters, accompanied by inverse thermal profiles. On another occasion, biogenic meromixis was again induced in these reservoirs by unusual meteorological conditions, when late winter ice cover and early summer thermal stratification prevented full vernal circulation. This allowed a building up of solutes in the bottom waters (Fiala, 1979).

Two cases of meromixis have been reported from reservoirs in the United States. Inflows of dense, turbid water created triptogenic

meromixis in Hills Creek Reservoir, Oregon (Larson, 1979), while in Lake Powell, on the Colorado River, cold, saline winter waters enter the reservoir as underflow currents to maintain ectogenic meromixis there (Johnston and Merritt, 1979).

2.6.3 Causes of Biogenic Meromixis in Reservoirs

The impoundment of heavily vegetated, steepsided river valleys may predispose the resultant reservoir to biogenic meromixis (Tyler, 1980). The valley shape concentrates debris onto the reservoir floor, so that even meagre epilimnetic production would produce a heavy sediment load per unit volume of bottom water, and this, along with the newly flooded, decaying vegetation, creates a high oxygen demand, which under conditions of thermal stratification causes an anoxic hypolimnion occupying much of the volume of the reservoir. This allows a large contact area between the anoxic water and the reservoir basin, from which iron and manganese can be reduced and mobilized. Being denser than the surrounds, water containing these, and other solutes of biogenic origin such as calcium, magnesium, and bicarbonate, will accumulate by downslope and down-reservoir migration to form a monimolimnetic pool, and thus establish a chemical gradient. The meromixis is localized in long, thin reservoirs, with the monimolimnion existing only as a small pool at the base of the dam (Tyler, 1980). A further factor aiding the maintenance of a chemical gradient is the shelter afforded by the steep valley sides and the dendritic nature of the reservoirs. This limits vertical turbulence caused by wind induced autumnal overturn and winter circulation, which would otherwise erode the monimolimnetic pool, and allows it to persist unless disturbed by low level offtakes or artificial aeration (Tyler, 1980).

CHAPTER THREE

STUDIES OF THE UNDERWATER LIGHT CLIMATES OF LENTIC FRESHWATERS FROM TASMANIA, NORTH-EAST NEW SOUTH WALES, AND SOUTH-EAST QUEENSLAND

3.1 INTRODUCTION

3.1.1 The Aims and Scope of the Study

The character of the underwater light climate is important not only biologically, in affecting primary production (Prakash et al, 1975; Salonen, 1984), the depth limitations of macrophytes (Vant et al, 1986), and possibly influencing the composition of phytoplankton communities by enabling different algae with specific pigment arrays to each have an ecological advantage somewhere in the range of light climates encountered in natural waters (Eloranta, 1978; Kirk, 1976b, 1979, 1981c, 1983; Jeffrey, 1980), but also in determining the optical quality of water and its suitability for domestic, industrial, and recreational use (Kirk, 1982, 1983; Davies-Colley, 1983; Vant and Davies-Colley, 1984). Thus, a knowledge of the optical properties of the aquatic medium, and the effects of the various attenuating components involved, are essential for an understanding of the functioning of inland aquatic ecosystems, for both biological and management purposes.

A necessary first step towards determining the ecological role of light in freshwater ecosystems is the characterization of their optical properties, and Kirk (1986) has stated that this task has scarcely begun for Australian inland waters. The major aim of this study was to characterize the optical properties of lentic freshwaters from several different areas of Australia, these being Tasmania, north-east New South Wales, and south-east Queensland. This was done by detailing the factors attenuating P.A.R. in these waters, and the

resultant light climates produced. Gilvin is an important light attenuating component of the aquatic medium, and emphasis has been made to determine its role in the optical limnology of the three study areas. Comparisons can also be made between the three areas, particularily as the types of waterbodies studied differ considerably from area to area.

A second aim of the studies undertaken in New South Wales and Queensland was to examine the distribution of phytoplankton in relation to the light climates of these study sites. Measurements of other ecological parameters, especially of the nutrients present, were also made, as possibly these, rather than the light climates, were the dominant factors affecting phytoplankton distribution. In doing so, both the nutrient status and the phytoplankton present in many of the freshwater coastal dune lakes of south-east Queensland have been documented for the first time, and this is also true for many of the north-east New South Wales study sites.

Many of Tasmania's phycologically rich lakes lie in remote areas, such as in the alpine regions of the Central Plateau, or in the World Heritage Wilderness Area of the south-west, where access is limited by rugged terrain and dense vegetation. A further aim of this study was the development of an optical classification of Tasmania's lakes, which could be used predictively to estimate the underwater light climates of the lakes from these areas, using only laboratory measurements of small water samples from them. Similar classifications were also done for north-east New South Wales and south-east Queensland study sites.

3.1.2 The Study Sites

Tasmania has a considerable range of lake types, many with water of high purity; many with dark, dystrophic waters, and a few which are turbid (Section 2.1). This study covers all water types, from as wide a geographic distribution of the island as possible, including one site on sub-Antarctic Macquarie Island. Basic limnological features of these lakes and reservoirs, with references to further data, are given in Table 3.1 and locations shown in Figure 3.1. Fifty sites were sampled, a number on more than one occasion.

Thirty-seven standing freshwaters were sampled in north-east New South Wales and adjacent south-east border regions of Queensland. A brief limnological background to the area is given in Section 2.2,

Table 3.1: Tasmanian lakes on which PAR measurements have been carried out, giving basic limnological data. The location of the lakes is given by lake number in Figure 3.1. Other numbers indicate the number of occasions when PAR measured. For additional general limnological data see Section 2.1.

Lake No.	Lake	Total PAR	Spectral distribution of PAR	Limnological characters and comments
Oligotrophic clear waters				
1	Prion	1		Macquarie Island, sub-antarctic. Tyler, 1972; Buckney & Tyler, 1974
2	Perry	2	1	Glacial, Hartz Mountains. Dolerite, Knott et al., 1978
3	Esperance	1		
4	Hartz	1		
5	Ladies Tarn	1		
6	Ridgeway Reservoir	1		Hobart water supply. Clear mountain water.
7	Fenton	1		Glacial, Mt Field. Dolerite
8	St Clair	1	1	Glacial. Cradle Mt - Lake St Clair National Park. Predominantly dolerite. Tyler, 1974; Buckney & Tyler, 1973a.
9	Laura	1	1	
10	Meston	1		
11	Junction	1		
12	Great	2	1	Central Plateau, alpine Dolerite, Tyler, 1974; Buckney & Tyler, 1973a.
13	Echo	1		Central Plateau. Dolerite. Buckney & Tyler, 1973a.
14	Lagoon of Islands (nova)	1	1	Sub-alpine. Central Plateau. Tyler, 1976.
15	Risdon Brook Reservoir	7	1	Hobart water supply. Treated water. Oligotrophic-mesotrophic. Turbidity variable. Tyler, 1974, 1980; Buckney & Tyler, 1973a.
16	Grants Lagoon	1		Coastal lagoon, north-east coast.
17	S.D. Marshall's Reservoir Ormley	1		Private reservoir, Fingal Valley.
18	Leake	1	1	Oligrtrophic reservoir, Eastern Tiers. Dolerite. Croome & Tyler, 1972, 1975; Buckney and Tyler, 1973a.
19	Trevallyn	1		Reservoir, near Launceston
20	Pet Reservoir	1		Town water supply, north-west coast, Basalt.
Dystrophic, non-turbid waters				
21	Binney	1		Alpine reservoirs. Central Plateau. Dolerite.
22	Pine Tier Dam	1		
23	King William	2		
24	Meadowbank	1		Reservoir, Lower Derwent Valley
25	Barrington	2	2	Slightly humic reservoir. Tyler & Buckney, 1974; Tyler, 1980.
26	Dove	1		Glacial, Cradle Mt - Lake St Clair National Park. Tyler, 1974; Buckney & Tyler, 1973a.
27	Curly	1	1	Glacial dystrophic. Denison Range. Precambrian conglomerates.
28	Wurrawina	1	1	
29	Diamond	1		
30	Rhona	1		
31	Pedder (nova)	3	2	Dystrophic reservoirs, south-west. Buckney & Tyler, 1973b; Steane & Tyler, 1982.
32	Gordon	3		
33	Murchison	2		Dystrophic reservoirs, west coast.
34	Rosebery	1		
35	Blackmans Lagoon	1		Coastal lagoon, north-east coast. Buckney & Tyler, 1973a.
36	Curries River Dam	1		Reservoirs, north-east. Buckney & Tyler, 1973a.
37	Frome Dam	1		
38	Cascades Dam (formerly Briseis Dam)	1		
39	Isandula	1		Dystrophic reservoirs, north-west coast.
40	Mikany	1		
41	Llewellyn	1		
42	Strahan	1	1	Dystrophic dune lakes, west coast.
43	Garcia	1		
44	Parting Creek Dam	1		Dystrophic reservoir, west coast.
45	Basin	1		Glacial dystrophic. Tyndall Range. Precambrian conglomerates.
46	Chisholm	4	1	Polyhumic forest lake. Karstic sink hole, Arthur River
Turbid waters (slight to moderate)				
47	Sorell	1	1	Meso-eutrophic, shallow. Central Plateau. Dolerite. Cheng & Tyler, 1973a, 1976a; Buckney & Tyler, 1973a.
48	Crescent	1	1	
49	Prosser River Dam	1		Slightly turbid reservoir, east coast.
50	Tooms	2	2	Oligotrophic, shallow reservoir. Eastern Tiers, dolerite. Croome & Tyler, 1972, 1975; Buckney & Tyler, 1973a.

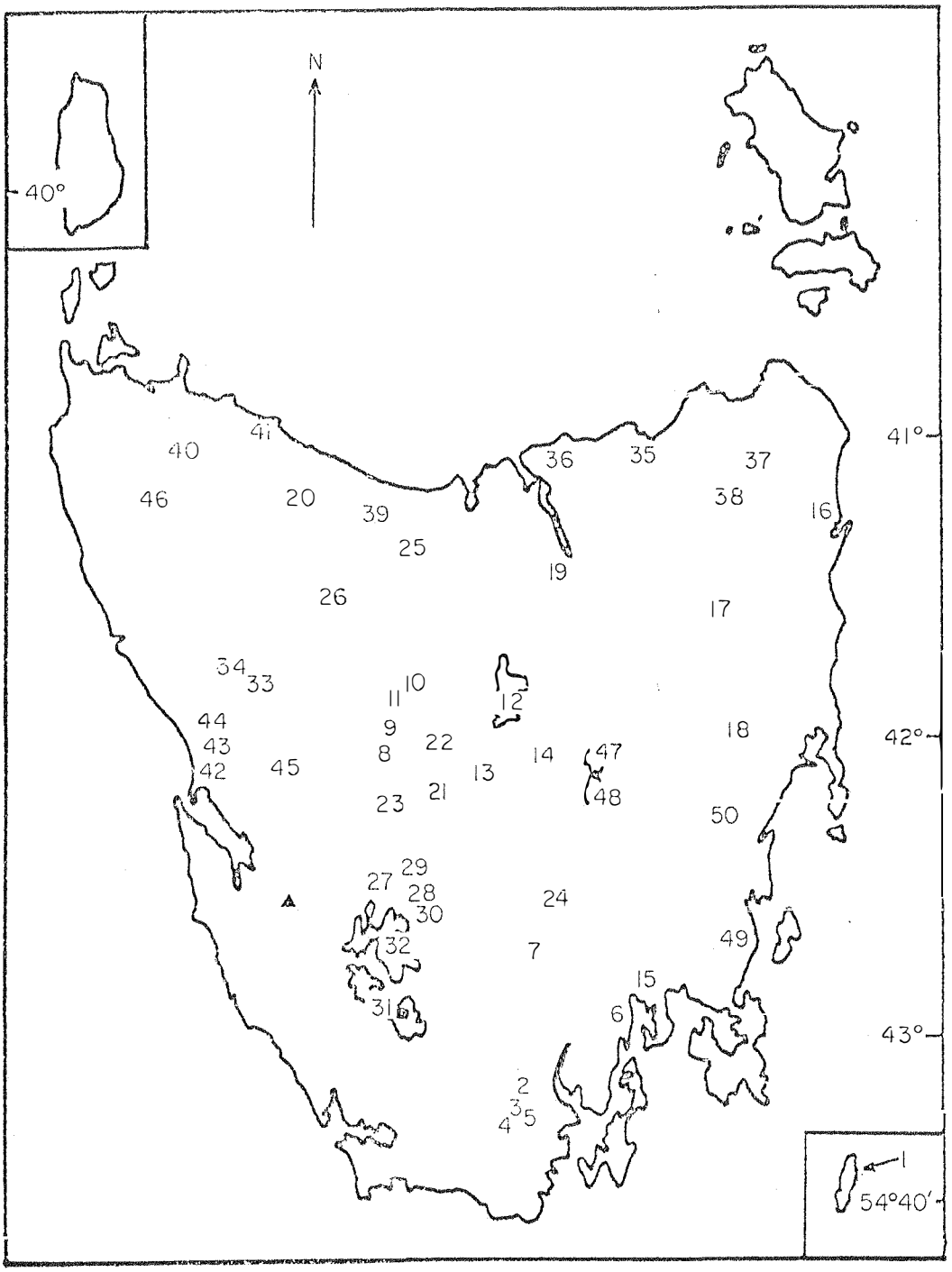


Figure 3.1 Location of sampling sites in Tasmania. The numbers refer to the lakes listed in Tables 3.1 and 3.2. Note the location of Prion Lake (No. 1) on sub-antarctic Macquarie Island. ▲ , Perched Lake, Lake Fidler, and Sulphide Pool, for which spectral distribution and attenuation data have been reported elsewhere. (King and Tyler, 1981b, 1982a, 1983; Croome and Tyler 1984a, 1985a; Bowling and Tyler, 1986).

which is one of topographic, geological, climatic and vegetational diversity. The eastern most section of the area includes the coastal plain and eastern foothills of the Great Dividing Range. Several coastal dune lakes, and a number of reservoirs, especially in the Richmond and Hunter River valleys, were investigated here (Nos. 1-10, 32-36, Table 3.4). The highland regions of the New England Tablelands comprise a second sector, containing many public water supply reservoirs and some private irrigation storages (Nos. 11-23, Table 3.4). Further west on the Western Slopes, are a number of large irrigation impoundments (Nos. 24-31, Table 3.4), while small, natural Yarrie Lake (No. 37, Table 3.4) is located on the Western Plains. Figure 3.2 shows the location of each site , which were sampled only once.

Twenty-six freshwater coastal dune lakes were sampled in south-east Queensland, including twenty from Fraser Island, and two each from the Cooloola Sand Mass, Moreton Island, and North Stradbroke Island (Figure 3.3, Table 3.8). The basic limnology of these lakes was described in Section 2.3.2. Because all have fairly similar thermal properties and ionic chemistry, but vary considerably in their humus content, they offer an excellent range of sites within a relatively small area on which to conduct investigations into the limnological effects of humic substances.

3.2 METHODS

3.2.1 Sources of Data

The majority of Tasmanian data for this study was collected between 1982 and 1984, inclusive. This was combined with additional data from other sources to obtain a sufficiently large data base to include all possible lake types from most geographical locations in Tasmania. This was important to allow comparisons both between individual lakes, and between those from different parts of the island. Data prior to 1978 was collected by P.A. Tyler (unpublished), while much of that for 1978 came from Steane (1979); the remainder being measurements by R.D. King and P.A. Tyler (unpublished). Data for 1979 was that of M.S. Steane (unpublished), while the results from Lake Strahan came from Bowling (1981). With the exception of that of Steane (1979) and Bowling (1981), all additional data was in the form of raw field measurements requiring computation to obtain the final results for each lake. As Steane (1979) had not applied a correction to account for the difference in refractive index of light

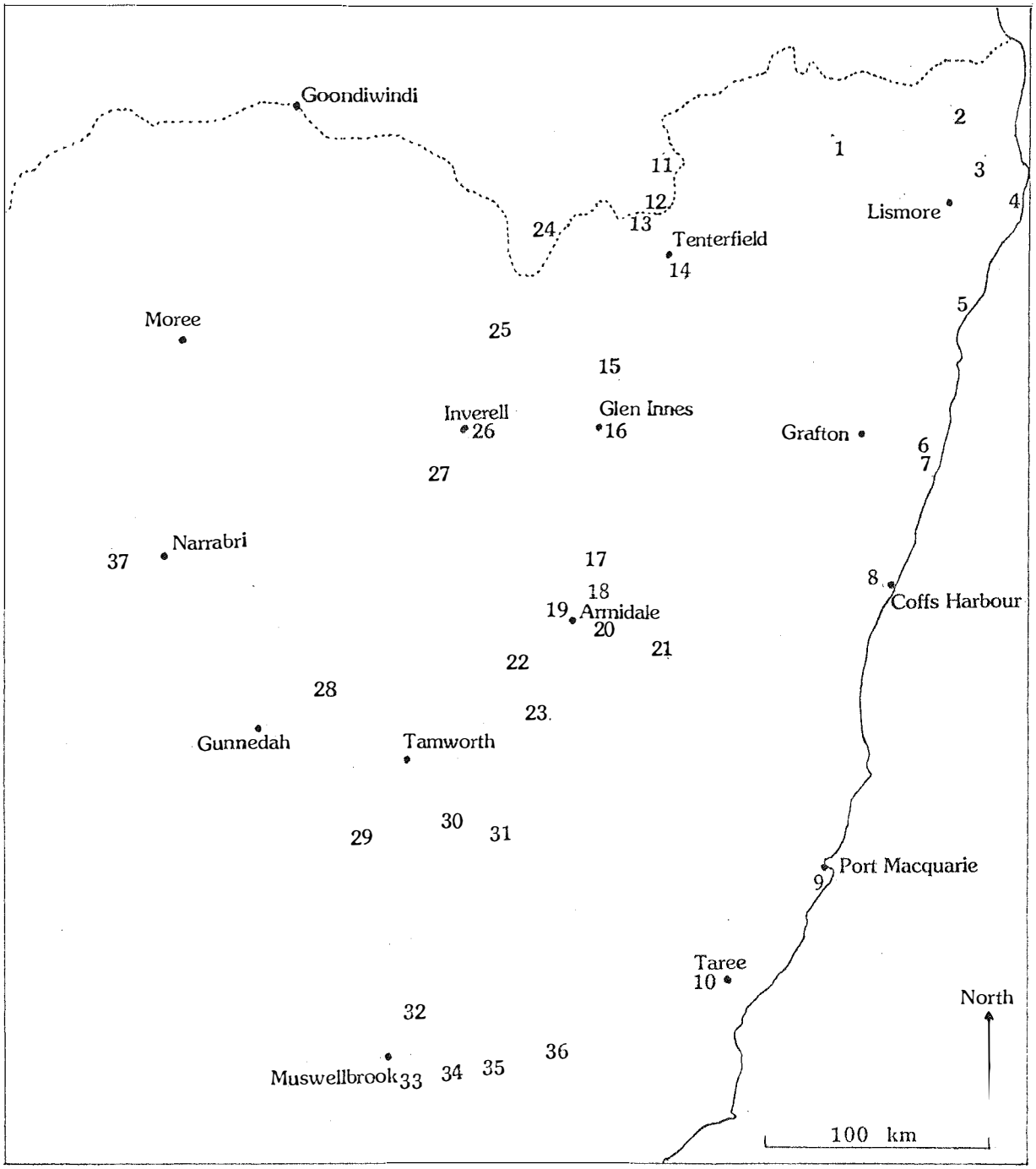


Figure 3.2 The location of sampling sites in north-east New South Wales. The numbers refer to the lakes and reservoirs listed in Tables 3.4 and 3.5.

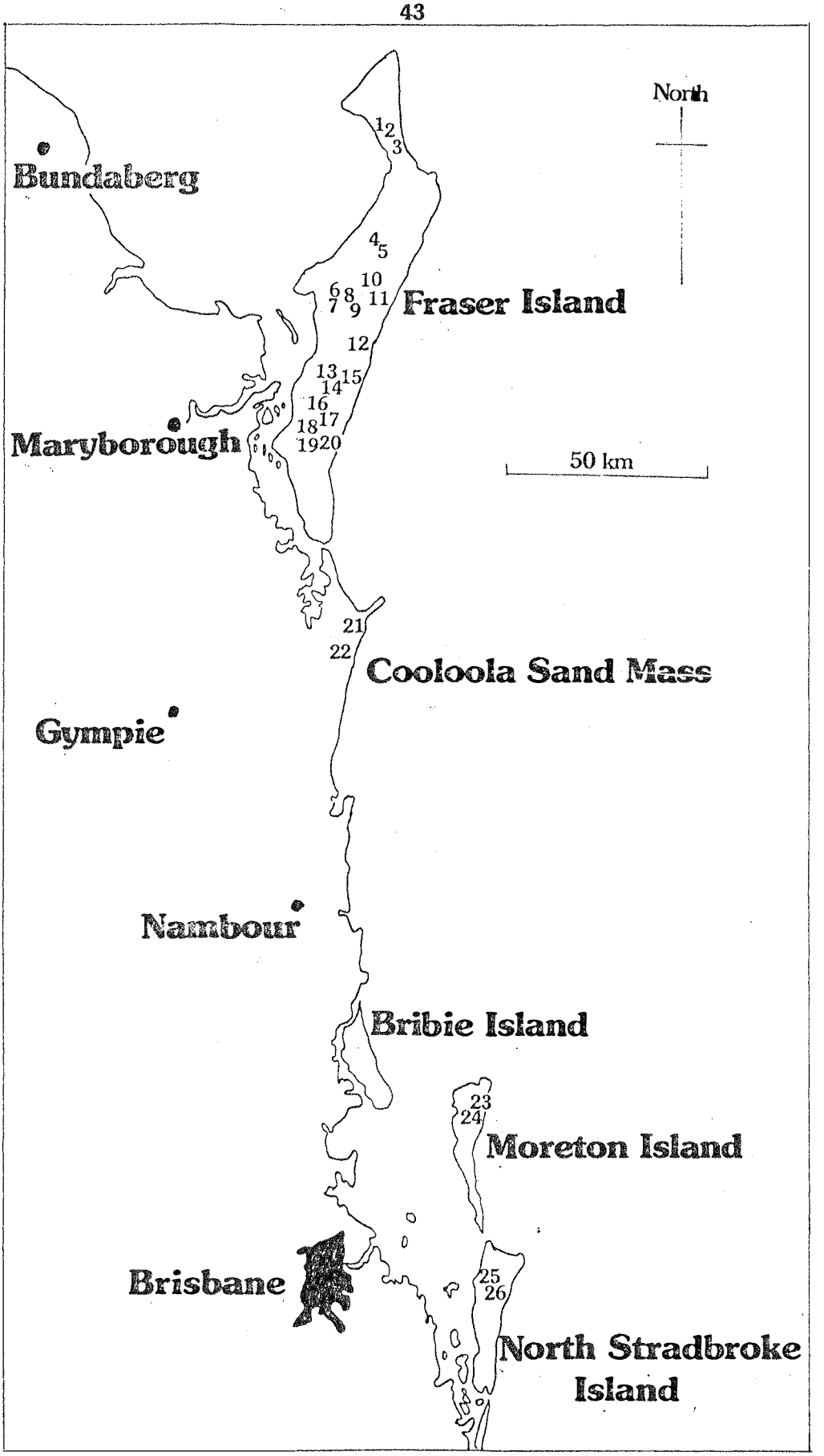


Figure 3.3 The location of sampling sites in south-east Queensland. The numbers refer to the lakes listed in Table 3.8.

in air compared to that in water, much of his data had to be recalculated before its inclusion in this study.

Data from north-east New South Wales was collected between July 1986 and September 1987; while that from south-east Queensland was obtained over a three week period from late April to early May, 1987.

3.2.2 Collection and Analysis of Samples, and Computation of the Data

Upwelling and downwelling measurements of P.A.R. (400-700 nm) were made with a Lambda Licor LI-185 quantameter and LI-192S underwater quanta sensor, always referenced to incident radiation measured simultaneously with a Licor LI-190S quanta sensor (deck cell) held horizontal above the lake surface. Measurements were usually made within three hours either side of solar noon, and, weather permitting, under clear skies. The depth of the first underwater measurement was always 0.10 metre, and the deepest measurements taken either at the euphotic depth or the bottom of the lake, whichever was the shallower.

From these measurements, mean upwelling and downwelling vertical attenuation coefficients for P.A.R. ($Ku_{(av)}$; $Kd_{(av)}$) were calculated by linear regression (Kirk, 1977b). The regression coefficients (r^2) for these were usually better than 0.98. These single, depth averaged values of Kd best typify the attenuation of P.A.R. within a given waterbody (Kirk, 1986), despite decreases in the rate of attenuation occurring due to the narrowing of the spectral distribution of P.A.R. with depth (Talling, 1982, 1984). Reflectance, R, was also calculated as the ratio of upwelling to downwelling irradiance at any depth (Kirk, 1977b). Euphotic depth, Zea, the depth of 1% of incident radiation, was determined directly from the measured downwelling P.A.R. profile. The scattering coefficient, $\underline{b}$, was calculated from upwelling and downwelling values of P.A.R. at the midpoint of the euphotic zone ($Zm \approx$ 10% of surface irradiance), following Kirk (1981a).

The spectral distribution of upwelling and downwelling P.A.R. from 400 to 700 nm, was measured with a Techtum Q.S.M. 2500 submersible quantaspectrometer (half-power bandwith = 15-18 nm). These were always made under a cloudless sky close to solar noon, under calm conditions. A slight rippling of the water surface was tolerable and under these conditions the output of the equipment was smoothed electronically with a device similar to the one described by Kirk (1979). Two scans were taken at each depth, one forward from 400 to 700 nm, and the second the reverse, these being recorded on a

constant speed chart recorder (TOA). There was always a slight discrepancy between scans in opposite directions. The start would be recorded correctly but the end of the scan would be indicated by the wavelength trace before the quantum trace had reached equilibrium. This error was overcome using the method of Steane (1979), where both mirror image scans, recorded in opposite directions, were superimposed in coincidence. The start of each scan was then marked as the finish of its partner. Remaining errors were minimized by scaling each scan and calculating means, at 10 nm intervals.

Approximate estimates of in situ total absorption coefficients were calculated at 10 nm intervals from the underwater spectral data for selected Tasmanian lakes, following Kirk (1979). This involves correcting the measured vertical attenuation coefficients for monochromatic downwards irradiance for solar altitude and refraction at the water surface. Absorption coefficients for pure water between 400 and 580 nm were taken from Hulbert (1945), and those from 580 to 740 from Sullivan (1963), while those attributable to gilvin were obtained from spectrophotometric scans of filtered water (0.45 µm membranes), measured relative to distilled water (Kirk, 1976b), in a Pye Unicam SP 8-100 UV/VIS scanning spectrophometer. The apparent contribution of tripton was then calculated as the difference between each total absorption coefficient and the sum of the coefficients for gilvin and water at that wavelength. However, because of the unavoidable contribution of scattering to attenuation of P.A.R., the values of both the total absorption coefficients and those for tripton, calculated by these methods, must exceed their true values to an extent dependant on the turbidity of the water (Kirk, 1979).

Transparency was measured with a standard 20 cm black and white quartered Secchi disc, surface water temperature with a mercury thermometer; and pH and conductivity (at 18°C) with a Jenco 673 pH meter and a PT1-10 conductivity meter respectively, the electrodes of each being suspended over the side of the boat into the water.

Samples of phytoplankton were collected by towing a 20 µm plankton net. These were preserved in 10% formalin, pending microscope identification, using Prescott (1970), McLeod (1975), Skinner (1975), Ling and Tyler (1986), Foged (1978) and Thomas (1983) as reference texts.

Samples for laboratory analysis were collected in opaque polyethylene bottles from below the surface. Turbidity (Tn) was measured in a Hach 2100 turbidimeter against formazan standards. Samples were then filtered through 0.45 μm membrane filters and colour (= gilvin) was measured in Hazen (mg Pt L^{-1}) units with a Lovibond 100 colour comparator (A.P.H.A., 1971), or as an absorption coefficient at 440 nm (g_{440}, ln units - Kirk, 1976b) with a Cecil CE 292 spectrophometer (Tasmania) or a Pye Unicam SP 30 spectrophometer. (New South Wales and Queensland samples) against a distilled water blank in 4 cm quartz glass cuvettes. These measurements were made as soon as possible after sampling. Additional water samples were filtered through Whatman GF/C glass fibre filters immediately after sampling. The filters were then stored frozen in darkness pending spectrophotometric analysis of phytoplankton chlorophylla by the trichromatic method of Strickland and Parsons (1968) following extraction in acetone (Tasmanian samples), or in methanol following the recommendations of Marker et al (1980) (N.S.W. and Queensland samples).

Further samples were collected for nutrient analyses in acid washed polyethylene bottles. Analyses were performed on a Technicon Autoanalyser II using the ascorbic acid method for total phosphorus, following acid persulphate digestion (A.P.H.A., 1971) (accuracy 2 µg l^{-1} Total P), and for total nitrogen after alkaline persulphate digestion (D'Elia et al, 1977) (accuracy 5 µg l^{-1} Total N).

Linear and multiple regression analyses of various optical characteristics of Tasmanian lakes were carried out using the Teddybear computer program, devised by J.B. Wilson, Otago University, with natural logarithmic transformations to correct for skewed and kurtosed distributions of the data. Similar analyses on data from New South Wales and Queensland were done using the Minitab program (Minitab Inc., 1985) for regressions, and SPSS (Nie et al, 1975) for Pearson correlations. Cluster analyses employing the Average Linkage Method with the parameters g_{440}, turbidity, and the g_{440} to turbidity ratio were done using the program Clustan 3 (Wishart, 1978).

3.3 RESULTS

PART A TASMANIA

3.3.1 Colour, Turbidity and Chlorophylla

Data for colour, turbidity and chlorophylla for the fifty lakes studied are given in Table 3.2. The least coloured water recorded was that of Lake Perry with a g_{440} of 0.058 m^{-1}, while the most dystrophic was Lake Chisholm, at 27.228 m^{-1}. The median value for Tasmania was 2.303 m^{-1}. The absorption coefficient at 440 nm (g_{440}) was taken as the precise measure of colour in this study. The relationship between this and the older, subjective method employing platinum units (Hazen scale) is shown in Figure 3.4. Hazen values were approximated to the nearest match in the incremental series of dyed discs, hence the vertical series of points at higher values of colour. Data from this study, and from studies of coastal lagoons (Table 4.1) were supplemented with additional data of King (1980). A linear relationship was found between the two colour measurements, expressed by the equation g_{440} = 0.081 Hazen + 0.40 (r^2 = 0.984, n = 320). King (1980) had demonstrated this relationship up to a Hazen value of 400 mg Pt L^{-1}, and inclusion of the extra data now extends it to at least 600 mg Pt L^{-1}. Similar relationships exist between the Hazen scale and spectrophotometric measures of colour at 420 nm for Finnish lakes (Eloranta, 1978; Jones and Arvola, 1984).

The lowest turbidity (0.18 N.T.U.) was recorded from Lake St. Clair, while the most turbid lake was Tooms Lake (17.0 N.T.U.). The median turbidity, 0.72 N.T.U., shows Tasmanian inland waters are generally less turbid than many of their mainland Australian counterparts (see Section 2.5.5.). Only eight values were recorded for chlorophyll, and six of these were below 10 μg L^{-1}. Lakes Sorell and Crescent are amongst the more eutrophic of Tasmania's lakes.

3.3.2 Attenuation of Total P.A.R.

(a) Downwelling P.A.R.

The attenuation of downwelling P.A.R. (400-700 nm) is shown for selected lakes in Figure 3.5. These cover the extremes of water clarity, turbidity, and dystrophy found in Tasmania. An increase in slope is apparent with depth in each profile, consequent upon the early eclipse of the spectral regions with high attenuation coefficients, in the surface waters. The mean values of the vertical attenuation

Table 3.2: Mean downwelling vertical attenuation coefficients for PAR ($Kd_{(av)}$), mean upwelling vertical attenuation coefficient for PAR ($Ku_{(av)}$) euphotic depths (Zeu, 1% downwelling), scattering coefficients (b), and related data for Tasmanian lakes. Figures in parentheses are ranges previously published (references in Table 3.1)

Lake No.	Lake	Date	Colour (mg Pt L^{-1})	a_{440} (m^{-1})	Turbidity (NTU)	Secchi (m)	Chloro-phyll ($\mu g\ L^{-1}$)	$Kd_{(av)}$ (m^{-1})	Zeu (m)	$Ku_{(av)}$ (m^{-1})	b (m^{-1})
Oligotrophic clear waters											
1	Prion	24-10-79	<5 (<5-5)	0.115	0.39	11.5	-	0.14	>10.0	0.09	0.60
2	Perry	10-04-78	<5	0.058	0.37	15.0	-	0.21	>16.0	-	-
		8-04-83	<5	0.058	0.20	15.0	-	0.21	20.0	0.18	0.27
3	Esperance	7-11-78	<5	0.173	0.27	16.8	-	0.25	18.0	0.30	0.47
4	Hartz	7-11-78	<5	0.115	0.28	9.1	-	0.25	18.0	0.23	0.41
5	Ladies Tarn	7-11-78	<5	0.403	0.26	>5.8	-	0.41	>5.5	0.18	-
6	Ridgeway Reservoir	9-06-83	<5	0.230	0.80	10.5	-	0.23	18.5	-	-
7	Fenton	14-11-82	<5	0.230	0.35	11.25	-	0.33	13.3	0.39	0.30
8	St. Clair	28-02-79	<5 (<5)	0.576	0.18	-	-	0.41	11.4	0.41	0.37
9	Laura	8-10-78	<5	0.748	0.22	8.8	-	0.43	10.7	0.41	0.38
10	Meston	23-02-78	<5	0.461	0.37	15.0	-	0.27	16.5	-	-
11	Junction	25-02-78	<5	0.806	0.38	>7.2	-	0.48	>6.0	-	-
12	Great	6-10-82	<5	0.230	0.50	>7.0	-	0.32	>6.5	0.23	0.76
		26-05-83	<5 (<5)	0.058	0.70	6.4	-	0.39	>7.5	0.35	0.98
13	Echo	4-02-84	<5 (<5)	0.230	1.00	6.2	-	0.37	12.0	-	-
14	Lagoon of Islands	15-07-82	10 (10)	1.036	0.75	>1.0	0.46	0.6	>0.9	0.41	-
15	Risdon Brook Reservoir	25-06-76	-	-	-	1.95	-	1.73	2.6	-	-
		2-11-78	15	0.979	1.5	5.3	-	0.6	7.5	0.62	1.78
		26-03-79	-	-	-	4.8	-	0.5	9.0	-	-
		4-05-81	-	0.230	4.4	1.4	4.64	1.2	3.5	-	-
		30-04-82	-	-	-	1.7	-	1.05	4.25	-	-
		1-05-82	<5	0.345	2.0	1.65	4.87	1.1	4.0	1.22	2.66
		15-06-84	5 (<5)	0.921	0.7	5.7	-	0.53	>5.0	-	-
16	Grants Lagoon	26-04-84	10	0.979	1.1	1.5	-	1.45	2.75	-	-
17	S.D. Marshall's Reservoir, Ormley	1-07-84	10	0.864	1.2	2.9	-	0.85	5.0	-	-
18	Leake	16-07-82	5 (<5)	1.209	1.4 (3.4-9.0)	>2.5 (1.4-3.3)	7.19	0.9	>2.0	0.9	1.76
19	Trevallyn	24-04-83	5	0.748	1.5	3.2	-	0.74	6.0	0.64	2.87
20	Pet Reservoir	15-03-84	<5	0.345	1.0	2.8	-	0.81	>3.0	-	-
Dystrophic non-turbid water											
21	Binney	5-07-84	15	1.439	2.0	3.35	-	0.99	4.3	-	-
22	Pine Tier Dam	5-07-84	10	1.612	1.2	6.4	-	0.65	>6.0	-	-
23	Lake King William										
	Main Arm	28-06-84	5	1.669	0.65	5.2	-	0.69	6.0	-	-
	Guelph Basin	28-06-84	20	2.188	1.0	3.0	-	1.09	4.0	-	-
24	Meadowbank	5-07-84	25	2.878	1.5	2.8	-	1.33	3.15	-	-
25	Barrington	13-03-78	20	2.188	1.55	3.1	-	-	-	-	-
		22-09-78	30	3.051	1.04	3.2	-	1.19	3.9	1.15	1.45
		2-10-82	35 (10-40)	3.339	0.50	3.3	-	1.27	3.5	1.63	1.13
26	Dove	3-10-82	35 (20)	3.339	0.47	4.9	-	1.03	4.2	1.04	0.75
27	Curly	22-01-78	65	6.056	0.80	2.25	-	1.94	2.3	-	-
28	Wurrawina	20-01-78	75	5.987	0.50	3.80	-	1.29	3.6	-	-
29	Diamond	17-01-78	90	7.093	0.30	3.10	-	1.75	>2.0	-	-
30	Rhona	17-01-78	65	4.905	0.40	2.90	-	1.77	2.6	-	-
31	Pedder (Nova)	26-08-75	-	-	-	2.4	-	2.3	2.0	-	-
		21-07-82	70	6.505	0.73	2.1	1.73	2.48	1.6	3.87	0.6
		27-06-84	90	8.174	0.38	2.2	-	2.35	1.7	3.82	1.32
32	Gordon	6-01-78	80	8.289	0.64	2.4	-	2.25	2.0	-	-
		3-11-78	85	7.023	0.32	2.55	-	1.94	2.2	2.58	0.98
		23-03-84	55 (80-100)	4.951 (6.2-8.3)	0.71 (0.30-0.65)	2.9 (2.05-2.95)	-	1.4	3.0	-	-
33	Murchison	8-11-82	80	6.678	0.65	2.70	-	1.76	2.5	-	-
		17-03-84	55	4.720	0.5	-	-	1.75	2.5	-	-
34	Rosebery	18-03-84	70	6.505	0.53	2.2	-	1.77	2.5	-	-
35	Blackmans Lagoon	25-04-84	65 (70)	4.490	0.9	>1.75	-	1.17	>1.7	-	-
36	Curries River Dam	4-06-83	60	4.433	1.60	3.1	-	1.13	3.8	1.24	2.44
37	Frome Dam	25-04-84	80 (40)	6.793	1.5	1.4	-	2.93	2.0	-	-
38	Cascades Dam	26-04-84	60 (20)	4.548	1.0	2.15	-	3.13	>1.0	-	-
39	Isandula	15-03-84	25	1.900	1.4	1.5	-	1.28	3.5	-	-
40	Mikany	16-03-84	30	2.418	1.3	1.75	-	1.11	4.0	-	-
41	Llewellyn	15-03-84	110	7.886	2.4	1.15	-	2.42	1.75	-	-
42	Strahan	16-01-81	120	11.628	0.4	0.7	1.9	4.5	>0.7	-	-
43	Garcia	12-08-84	40	3.799	0.5	3.3	-	2.14	2.0	-	-
44	Parting Creek Dam	12-08-84	180	16.464	0.85	1.2	-	4.28	1.0	-	-
45	Basin	10-08-84	125	10.707	0.51	1.6	-	3.25	1.25	-	-
46	Chisholm	18-02-84	320	27.228	0.43	0.8	-	4.88	0.8	-	-
		16-04-84	300	24.695	0.43	0.95	-	4.51	0.9	-	-
		10-06-84	300	24.235	0.75	1.10	-	5.37	0.7	-	-
		13-08-84	320	26.192	0.55	1.05	-	3.82	1.0	-	-
Turbid waters (slight to moderate)											
47	Sorell	14-07-82	5 (<5)	1.209	4.3	1.0 (0.1-1.2)	26.61	1.8	>1.8	1.82	7.96
48	Crescent	15-07-82	15 (<5)	1.727	9.4	0.6 (0.1-0.6)	28.06	3.96	1.1	3.76	15.17
49	Prosser River Dam	25-02-84	60	5.699	7.1	1.1	-	1.75	2.5	-	-
50	Tooms	22-06-78	45	4.030	17.0	1.2	-	2.16	2.0	-	-
		31-08-82	45 (<5)	4.663	12.0 (10.0-14.0)	1.1 (1.5-2.1)	-	2.15	2.1	2.12	10.94
	MEDIAN		25	2.303	0.72	2.65	4.64	1.20	3.50	0.64	0.98
	MEAN		53	4.629	1.55	4.06	9.43	1.55	5.11	1.60	2.81
	STANDARD DEVIATION		77	6.292	2.79	3.94	11.26	1.24	5.14	1.99	3.88

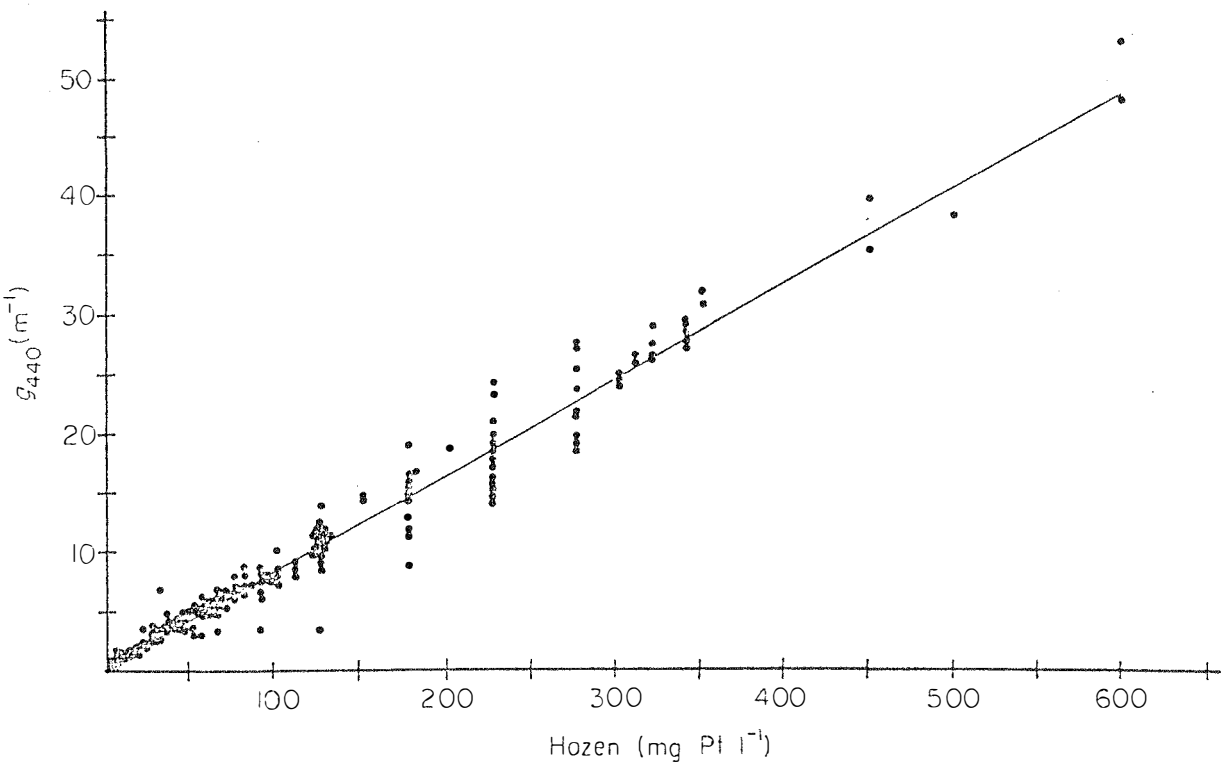


Figure 3.4 The relationship between colour measured by the platinum (Hazen) scale, and as gilvin (g_{440}) for Tasmanian inland waters.

coefficient, $Kd_{(av)}$, obtained from a line force-fitted by regression analysis, are given in Table 3.2, and are used to typify the underwater light climates of Tasmanian lakes. Euphotic depth (Zeu) is also given in Table 3.2. The median values for Kd(av) and Zeu were 1.20 m^{-1}, and 3.5 metres, respectively.

Table 3.2 and Figure 3.5 reveal that both turbidity and colour markedly increase the vertical attenuation coefficient, $Kd_{(av)}$. The highest values (greatest attenuation and shallowest euphotic depths) were measured in the turbid, eutrophic, but low coloured Lake Crescent, and from Lake Chisholm, a non-turbid, polyhumic forest lake. The least attenuation was in the very oligotrophic glacial lakes on dolerite bedrock, such as Lakes Perry, Laura, and Meston, and in Prion Lake on Macquarie Island. Midway were the moderately dystrophic Lake Barrington and the slightly turbid Lake Leake and Risdon Brook Dam. Attenuation in Lake Crescent was slightly higher than expected for its turbidity, especially when compared with the more turbid Tooms Lake, possibly due to additional absorption by chlorophylla.

(b) Upwelling P.A.R.

In most lakes, the mean attenuation coefficient for upwelling P.A.R. ($Ku_{(av)}$) was approximately equal to that for downwelling P.A.R. (Table 3.2). Notable exceptions were Ladies Tarn and Lagoon of Islands, where $Ku_{(av)}$ was considerably less than Kd(av). Both are shallow, with almost 10% of surface irradiance reaching the sediments. Steane (1979) attributed the differences in the $Ku_{(av)}$ and Kd(av) values from Ladies Tarn to a contribution to upwelling light by reflection from the bottom. This may also be true for Lagoon of Islands. In contrast, attenuation of upwelling irradiance in humic, non-turbid Lakes Gordon and Pedder (nova) exceeded that for downwelling irradiance. This is most likely the result of poor instrumental accuracy at the very low levels of upwelling irradiance present in these lakes.

3.3.3 Reflectance and Scattering Coefficients

Reflectance profiles from eight of the lakes are presented in Figure 3.6. The predicted pattern of reflectance becoming constant with depth after an initial increase (Kirk, 1977b, 1981a) does not appear to hold for Tasmanian lakes. In most, reflectance first increased with depth but then levelled off before decreasing. The

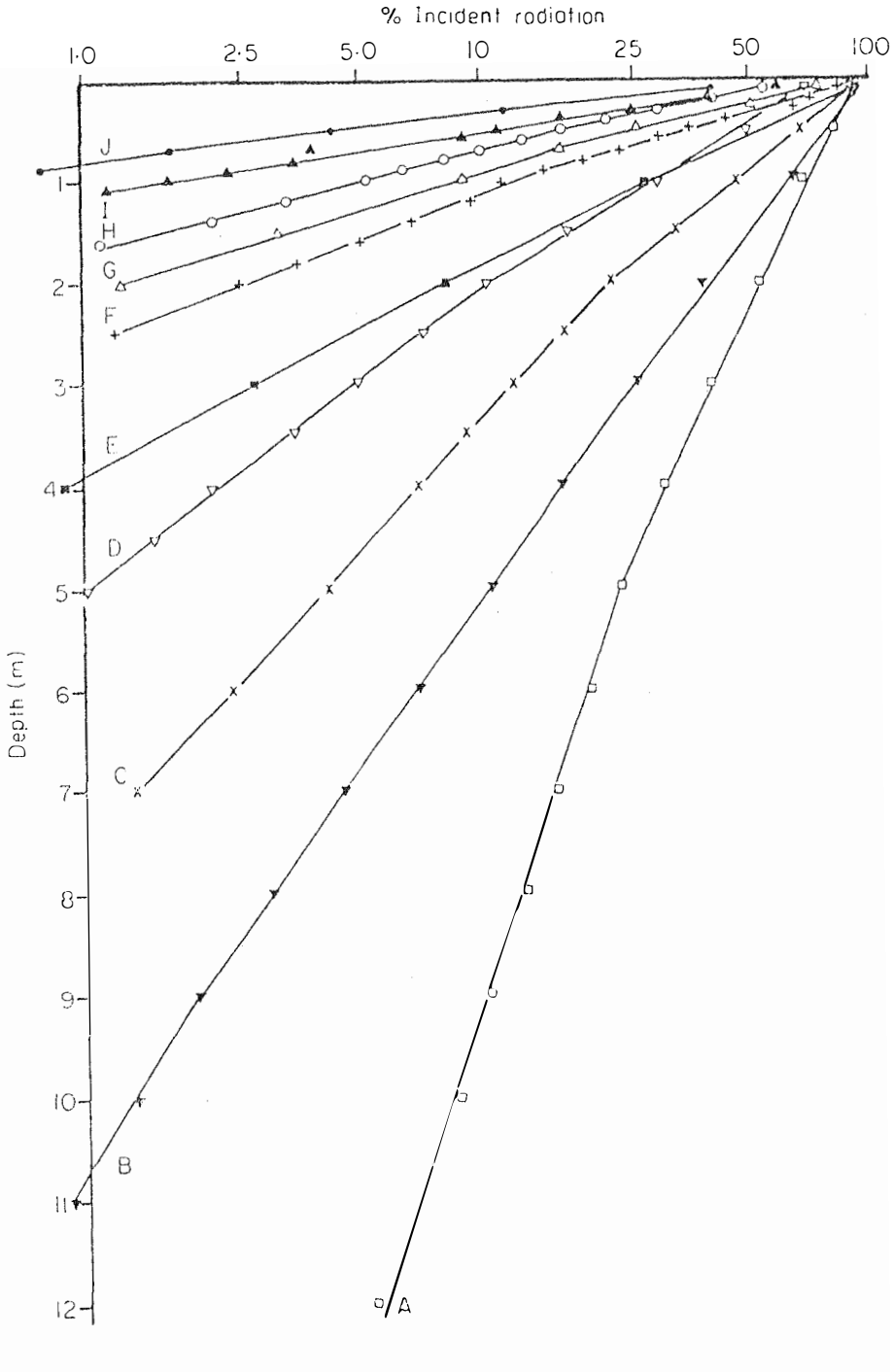


Figure 3.5 Attenuation profiles of downwelling P.A.R. (400-700 nm.) for selected Tasmanian waters. (A) Lake Perry, 08-04-1983; (b) Lake Laura, 08-10-1978; (C) Risdon Brook Dam, 02-11-1978; (D) "S.D. Marshalls' Reservoir", 01-07-1984; (E) Lake Barrington, 22-09-1978; (F) Lake Rhona, 17-01-1978; (G) Tooms Lake, 31-08-1982; (H) Lake Pedder (nova), 22-07-1982; (I) Lake Crescent, 15-07-1982; (J) Lake Chisholm, 18-02-1984. See Table 3.2 for further data on the optical characteristics of these lakes.

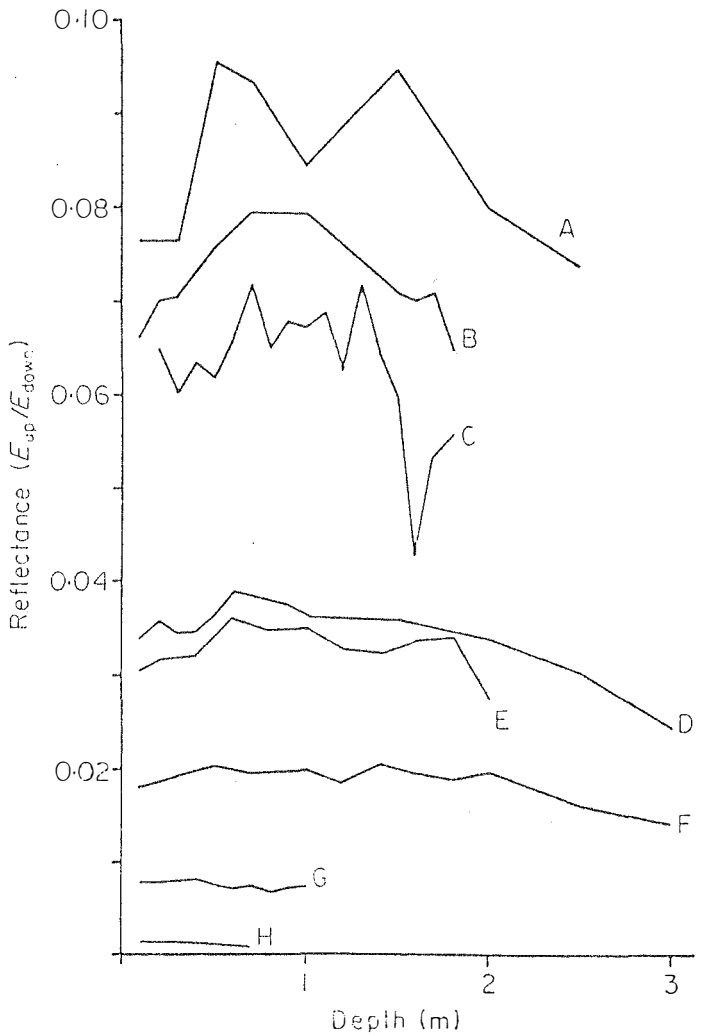


Figure 3.6 Reflectance/depth profiles for selected Tasmanian lakes. (A) Tooms Lake, 31-08-1982; (B) Lake Sorell, 14-07-1982; (C) Lake Crescent, 15-07-1982; (D) Risdon Brook Dam, 01-05-1982; (E) Lake Leake, 16-07-1982; (F) Curries River Dam, 04-06-1983; (G) Lake Barrington, 02-10-1982; (H) Lake Pedder (nova), 21-07-1982.

reasons for the highly variable reflectance profiles of Lake Crescent and Tooms Lake cannot be explained. The measurements were made under almost ideal conditions of cloudlessness and calm water, and instrumental malfunctions are unlikely. However, possibility of small errors in positioning the sensors at various depths during measurement, and vertical heterogeneity of phytoplankton and tripton during calm periods of measurement, may account for some variation.

The low quantities of tripton caused both the clear and the humic lakes to have low reflectance values, with upwelling P.A.R. usually being less than 1.5% of downwelling values. Exceptions were the more turbid lakes (Tooms, Sorell, Crescent), where upwelling P.A.R. ranged from 5% to 10% of the downwelling measurements. As a result, the scattering coefficients (b) for most Tasmanian lakes (Table 3.2) were also low, being usually less than 3.0 m^{-1}, except for Lakes Sorell and Crescent and in Tooms Lake, where, due to the greater turbidity, they were up to five times higher.

3.3.4 Spectral Distribution of Underwater P.A.R.

(a) Underwater spectra

The underwater spectral distributions of P.A.R. (400-740 nm) for various Tasmanian lakes are given in Figure 3.7 (a-e). Figure 3.7(a) shows the results for the four lakes where attenuation of P.A.R. was least. In Lake Perry, the clearest, red light was attenuated slightly more rapidly than blue light, leaving a spectral distribution centred on the green wavelengths, around 560 nm. In the other three lakes, attenuation of both blue and red wavelengths was almost equal, leaving orange light, centred around 580 nm, penetrating to the greatest depths. This spectral shift towards the red end of the spectrum indicates that very small quantities of gilvin or tripton cause perceptible changes in the underwater light climates. As turbidity, gilvin, or both increase, so does attenuation of blue light relative to red, so that the centre of the spectral distribution moves towards the longer wavelengths. This tendancy is further illustrated by Lagoon of Islands, with slightly increased colour; Lake Leake, and slightly turbid but almost colourless Risdon Brook Dam (Figure 3.7b). These waterbodies display a rather broad plateau in their underwater spectra, but with maximum transmission centred around 600 nm.

The effects of increased turbidity are seen in Lakes Sorell, Crescent and Tooms (Figure 3.7c), where the spectral bandpass has been

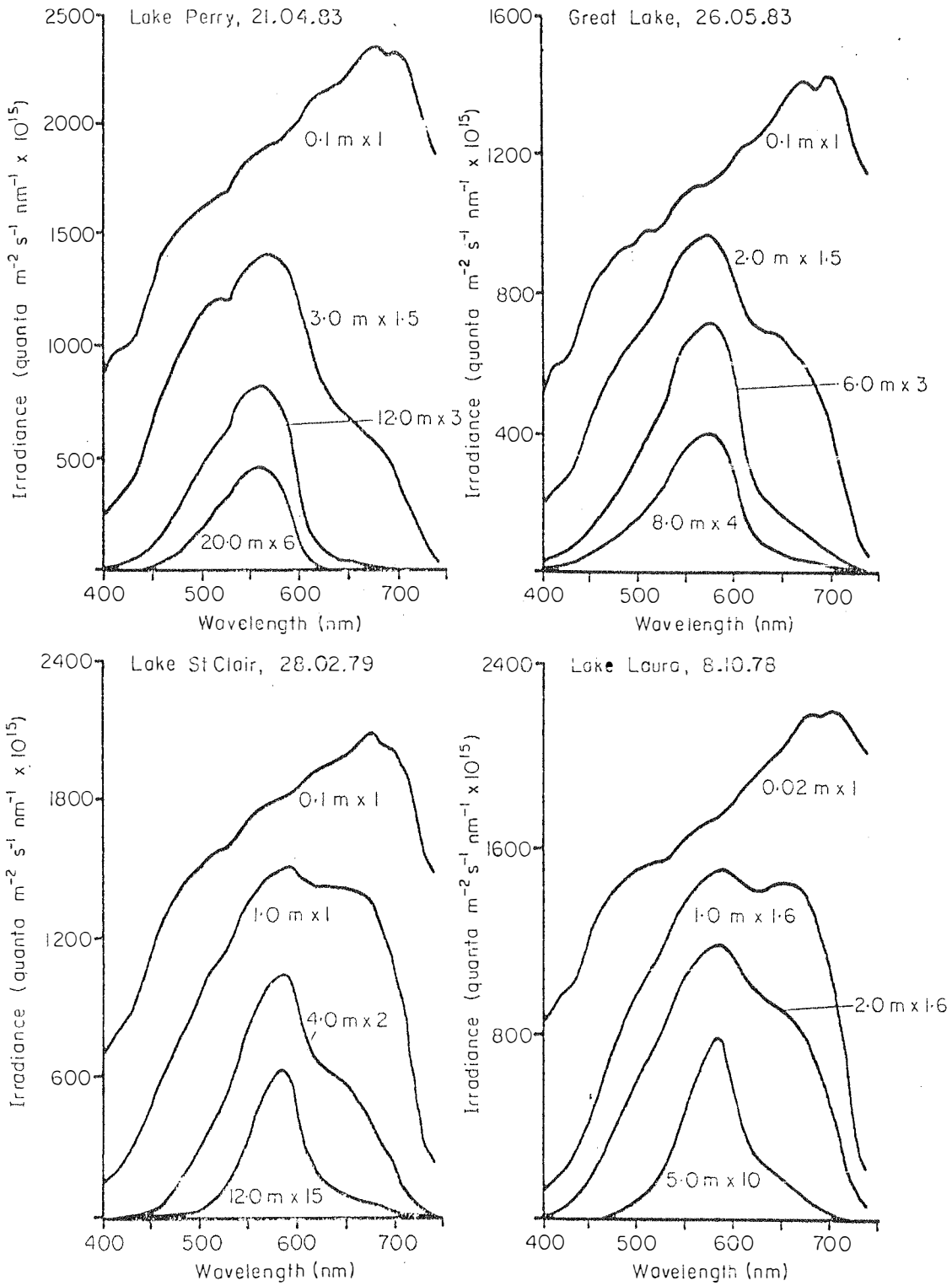


Figure 3.7(a) Spectral distributions of downwelling P.A.R. from Tasmanian lakes - clear water lakes.

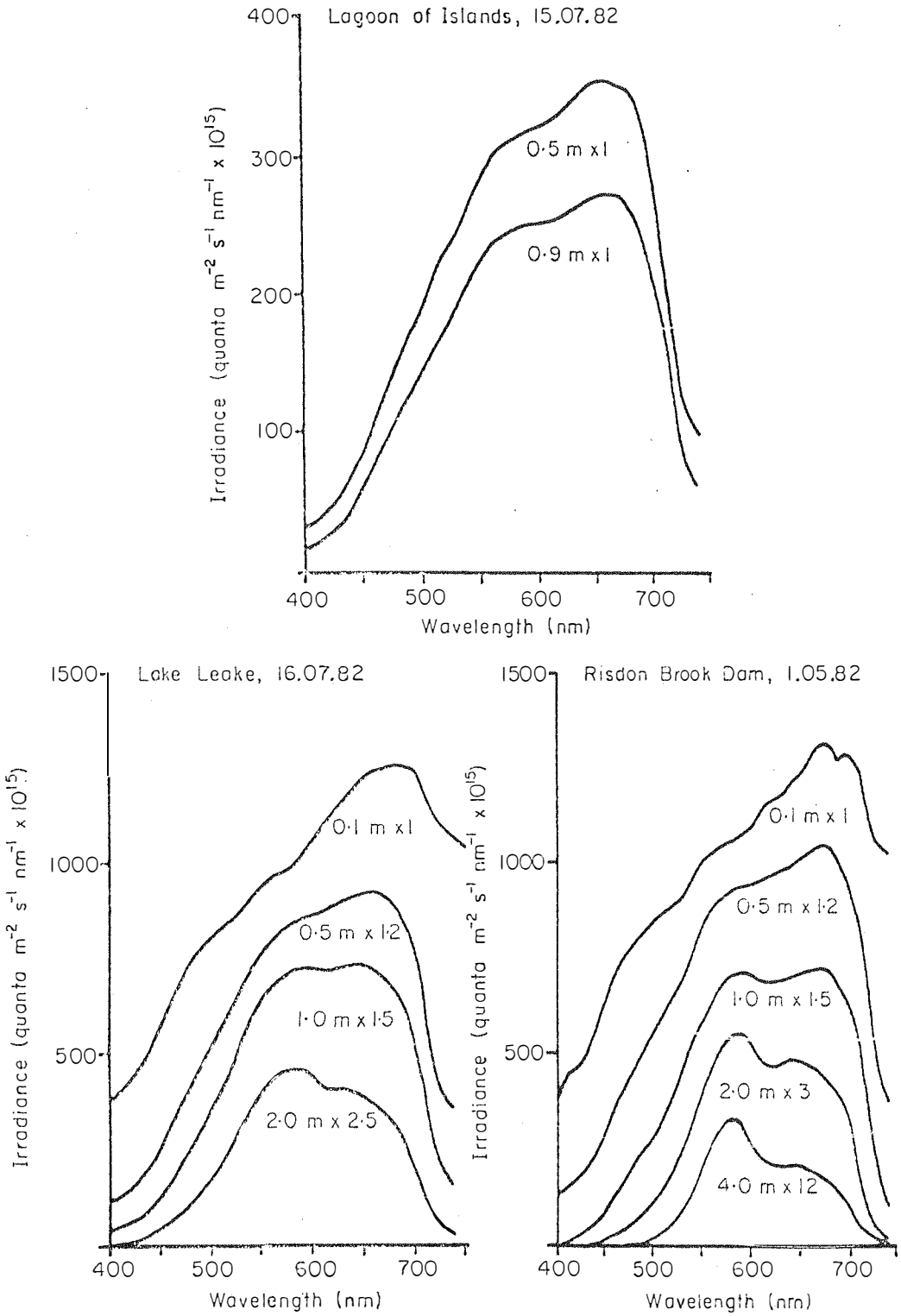


Figure 3.7(b) Spectral distributions of downwelling P.A.R. from Tasmanian lakes - slightly coloured or turbid lakes.

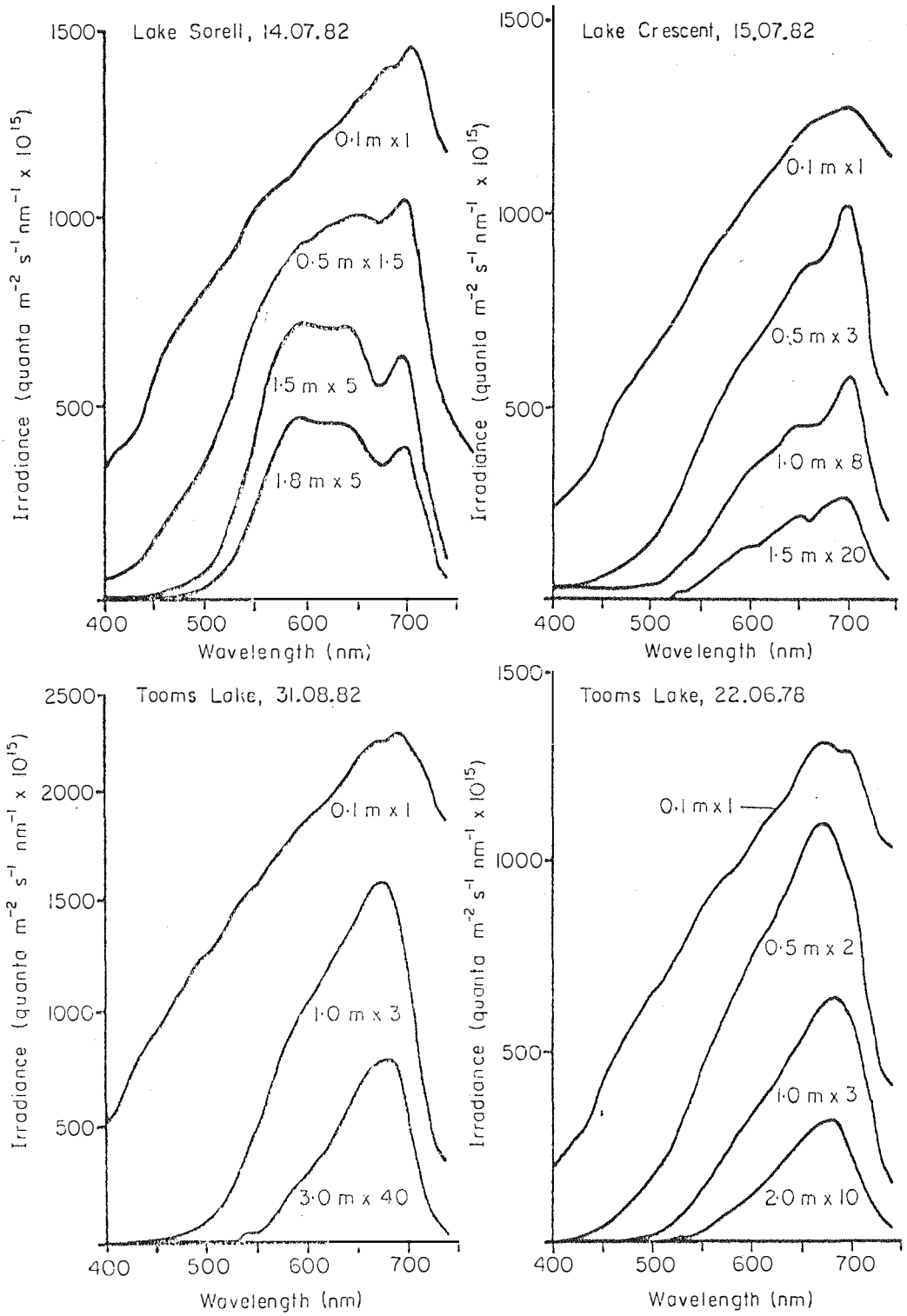


Figure 3.7(c) Spectral distributions of downwelling P.A.R. from Tasmanian lakes - turbid lakes.

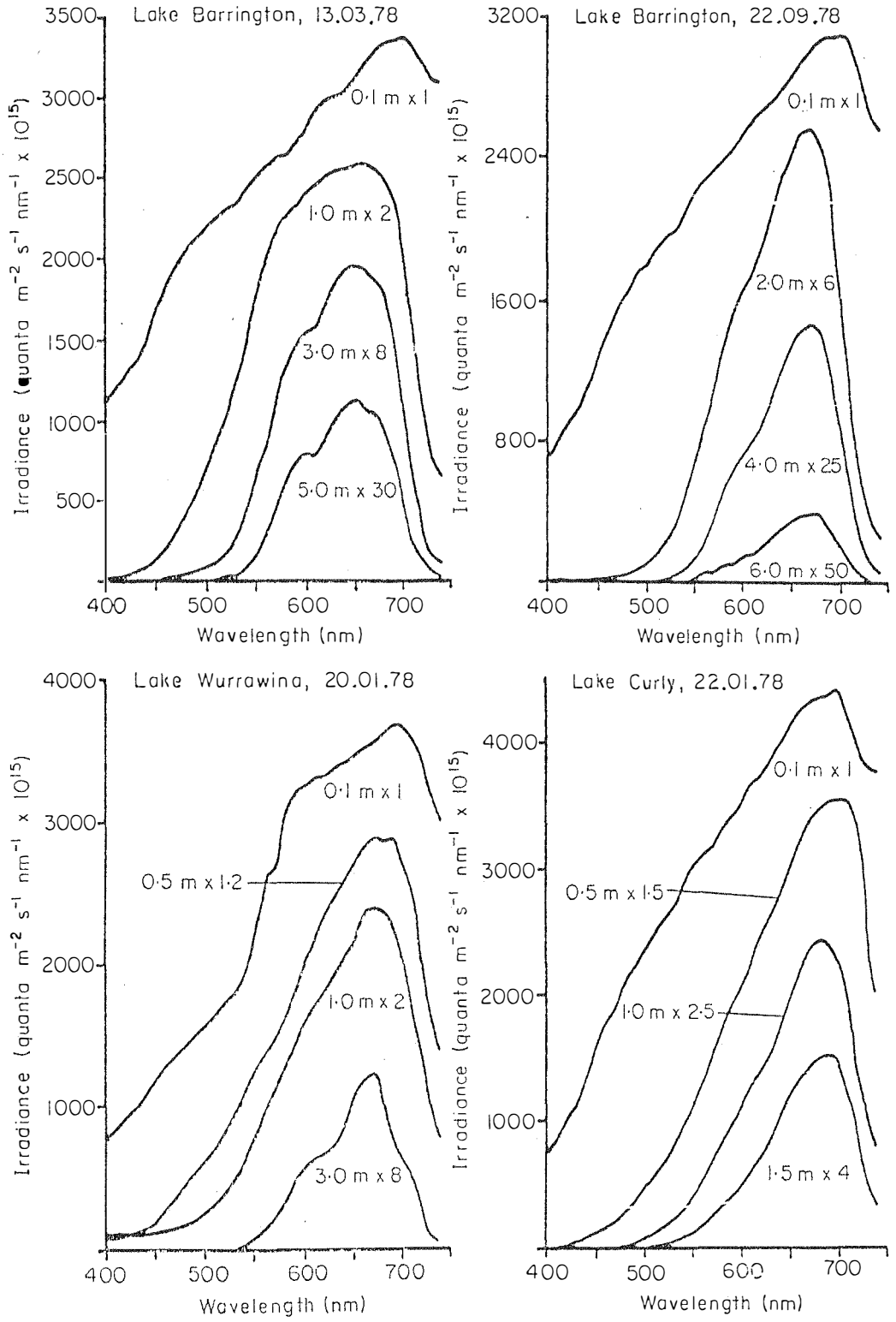


Figure 3.7(d) Spectral distributions of downwelling P.A.R. from Tasmanian lakes - moderately dystrophic lakes.

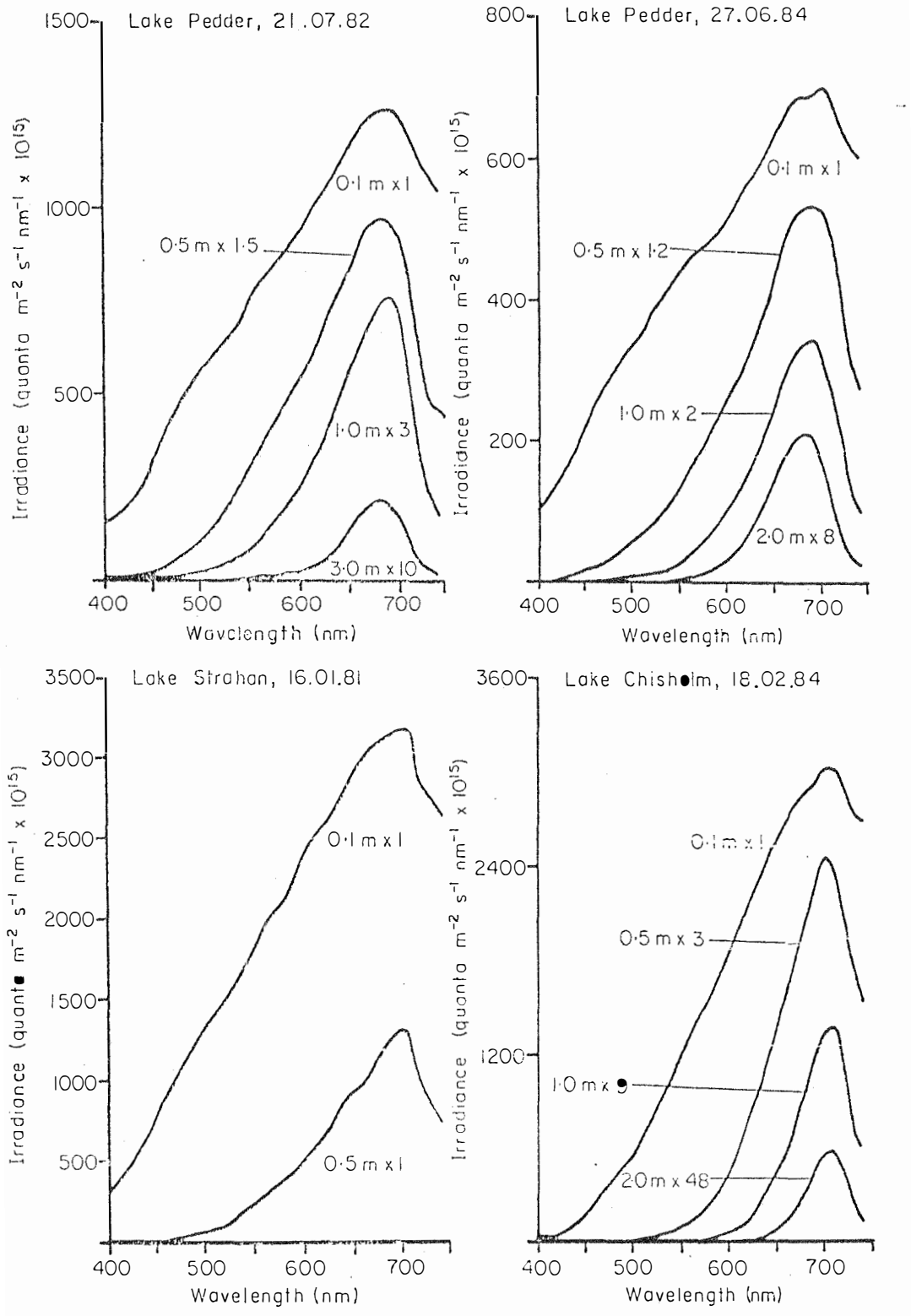


Figure 3.7(e) Spectral distributions of downwelling P.A.R. from Tasmanian lakes - highly dystrophic lakes.

further narrowed and moved to the right. Lake Sorell had low concentrations of gilvin, but is eutrophic, with its moderate turbidity caused in part by phytoplankton, predominantly Bacillariophyceae and Chlorophyceae (Cheng and Tyler, 1973a). Blue light is quickly attenuated, leaving a transmission plateau between about 580 to 700 nm. This is interupted by a trough centred between 670 and 680 nm, caused by the absorption of these wavelengths by chlorophylla contained in the phytoplankton in the water. In more turbid Tooms Lake and Lake Crescent the plateau is absent, and the transmission peak is close to 700 nm. A slight shoulder, possibly due to chlorophylla, is present at about 660 nm in the scans from Lake Crescent. In these turbid lakes, all wavelengths less than about 500 nm are rapidly extinguished, in the case of Lake Crescent in the first 1.5 metres.

The presence of dissolved humic materials in non-turbid, dystrophic lakes produces an underwater light climate very similar to that of colourless, turbid lakes. The spectral distributions of light at various depths are shown in Figure 3.7d for mesohumic Lakes Barrington, Wurrawina and Curly; and in Figure 3.7e for the more dystrophic Lakes Pedder (nova) and Strahan, and for polyhumic Lake Chisholm, which had the highest gilvin content of all Tasmanian lakes studied. Wavelengths below about 500 nm are quickly removed from the downwelling P.A.R., with the wavelength of maximum transmission lying at the red end of the spectrum, between 650 and 700 nm. The more humic the water, the further the transmission peak lies towards the longer wavelengths.

The spectral distributions of upwelling P.A.R. were measured in one of each of the three principal types of lake - clear, humic or turbid. These are shown in Figure 3.8. They were little different from the downwelling patterns at depth for the same lakes, apart from the greatly reduced amplitude. The paucity of upwelling irradiance in Lake Pedder, compared with the other two lakes is notable.

(b) Derived Spectral Coefficients

Kirk's (1979) method for comparing the extent to which each component of the aquatic medium contributes to the extinction of underwater light was applied to six Tasmanian lakes, including examples with clear, humic, or turbid waters (Figure 3.9). Particulate matter plays very little part in attenuation, especially of shorter wavelengths, in the state's oligotrophic and dystrophic lakes (e.g. Perry, Curly),

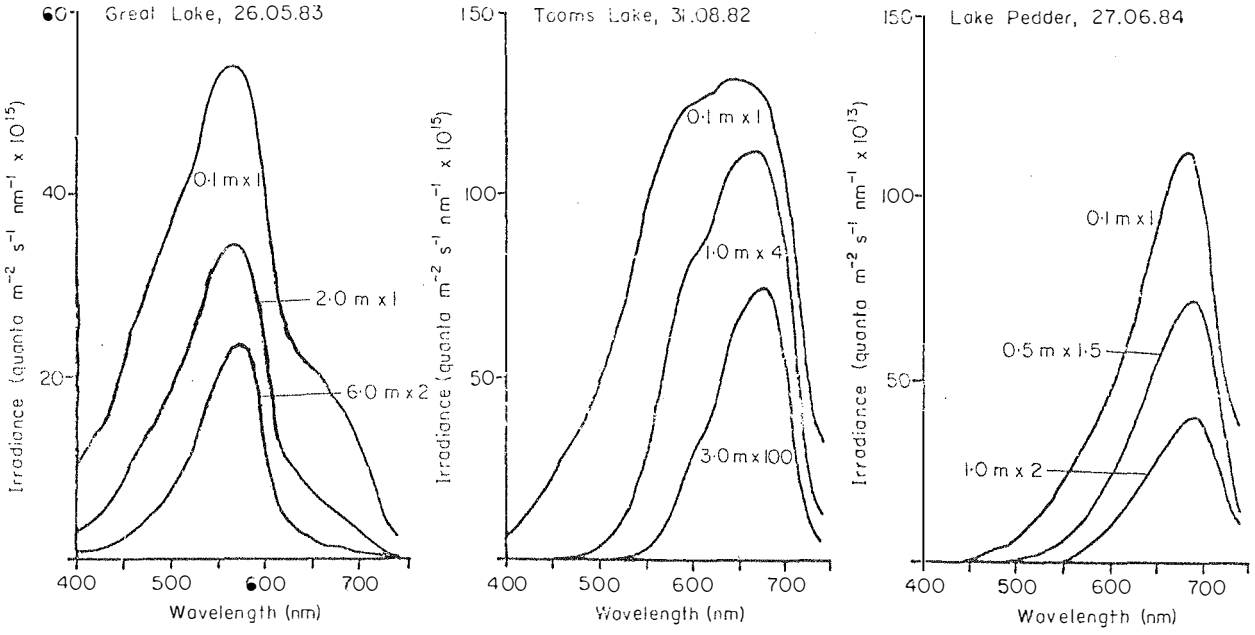


Figure 3.8 The spectral distribution of upwelling P.A.R. in clear, turbid, and dystrophic Tasmanian lakes respectively. Note that the irradiance scale for Lake Pedder (nova) is 10^{-2} that of the other two lakes.

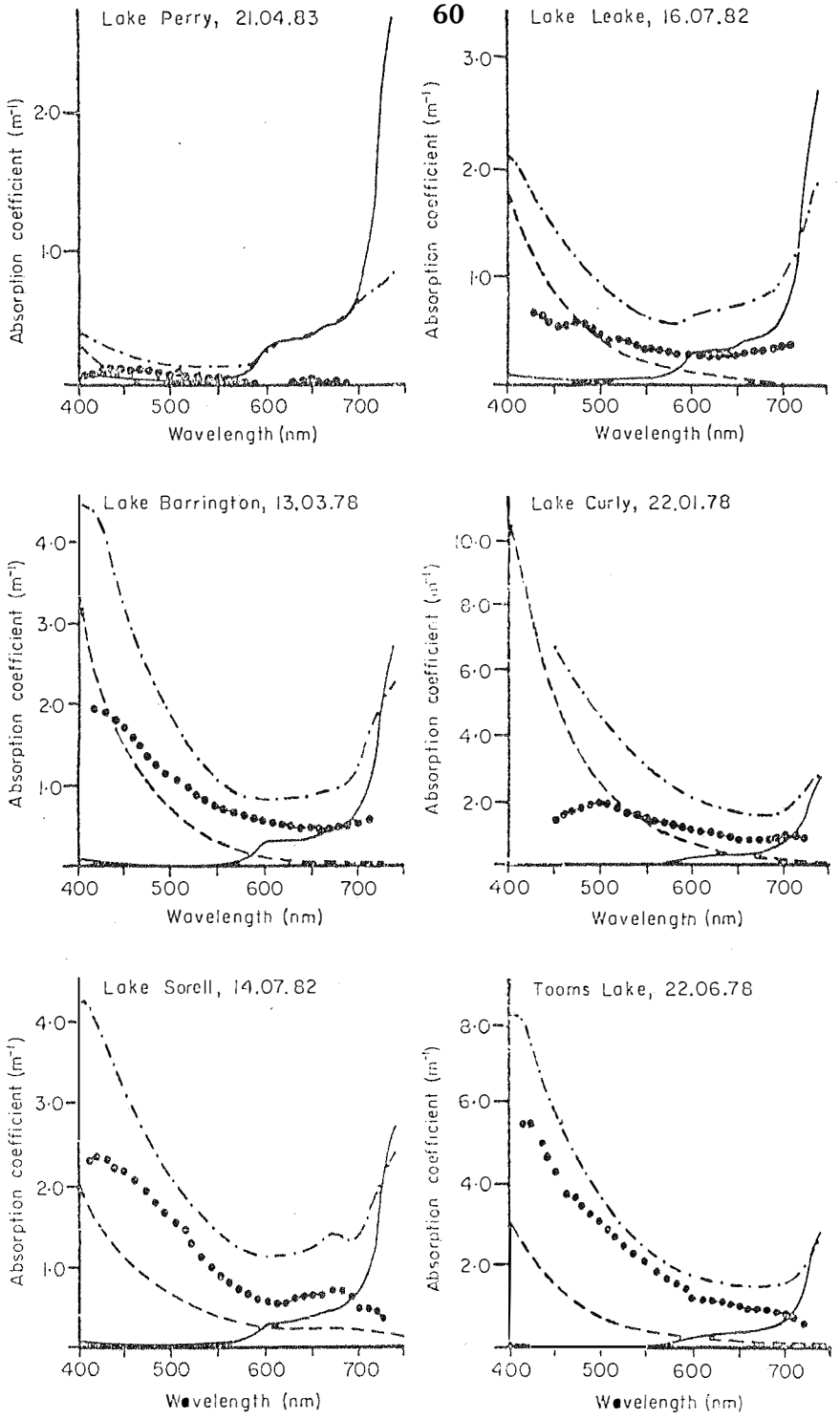


Figure 3.9 Comparison of calculated absorption coefficients at 10 nm. intervals over the P.A.R. spectrum for selected Tasmanian lakes. _____, Absorption coefficient due to water. ------, Absorption coefficient due to gilvin. ·-·-·-·-·-, The apparent total absorption coefficient obtained by quantaspectrophotometry within the lakes. ..... , The difference between the apparent total absorption coefficient and the sum of those for gilvin and water, giving an estimation of the contribution of tripton to total absorption. Some of this contribution must also be due to the scattering effect of tripton.

where either gilvin or water are the dominant extinguishers of light; but in the slightly more turbid lakes (e.g. Barrington, Leake), there is, naturally, a greater contribution, which may exceed that of the other components. Particulate matter is the dominant influence, however, in the turbid Tooms Lake and Lake Sorell, with a contribution from chlorophyll in the latter, shown by the increased absorption coefficients calculated for tripton from 660 to 680 nm.

However, it must be stressed that this method is approximate, providing a guide only. This is especially so where irradiance values are too low to measure accurately, such as below about 450 nm, causing spurious apparent total absorption coefficients for these wavelengths in the turbid or more dystrophic lakes. Kirk (1979) discussed shortcomings of the technique for regions of rapid change in attenuance, such as at the red end of the spectrum.

3.3.5 Secchi Disc Transparency

Secchi disc depths (Z_{SD}) for each lake are also presented in Table 3.2. Secchi disc depths were greatest in the clearest lakes, but decreased markedly with increased gilvin or turbidity. The maximum depth measured, 16.8 metres, was in Lake Esperance, and the minimum, 0.60, in Lake Crescent. The median Secchi disc depth for Tasmanian freshwaters was 2.65 metres.

3.3.6 Regression Analyses

Regression analyses (Table 3.3) shows the mean vertical attenuation coefficient for downwelling irradiance, $Kd_{(av)}$, is greatly influenced by gilvin concentration, with a statistically satisfactory regression linking the two. In comparison, turbidity measurements are poor predictors of attenuation in Tasmanian lakes with the regression $Kd_{(av)}$ against Tn being non-significant at the 5% level, although natural logarithmic transformation improved this considerably. The multiple regressions of Tn and g_{440} against $Kd_{(av)}$ do little to improve on those between g_{440} and $Kd_{(av)}$ alone. There was no correlation between g_{440} and turbidity in Tasmanian lakes ($r = -0.072$).

Although a highly significant linear relationship exists between Secchi disc depth and euphotic depth, Secchi disc measurements generally underestimate Zeu for many clear-water Tasmanian lakes and over-estimate it for dystrophic waters. In many instances the Secchi disc depth actually exceeded the euphotic depth measured by the quantaradiometer in dystrophic waters (Table 3.2).

Table 3.3: Simple and multiple regression analysis showing the relationships between various optical characteristics of Tasmanian lakes

Equation No.		r^2	n	P
1	$Kd_{(av)} = 0.746 + 0.173\ g_{440}$	0.747	65	$\ll$ 0.001
2	$\ln Kd_{(av)} = 0.508\ \ln g_{440} - 0.215$	0.806	65	$\ll$ 0.001
3	$Kd_{(av)} = 1.440 + 0.074\ Tn$	0.027	65	0.195
4	$\ln Kd_{(av)} = 0.149 + 0.313\ \ln Tn$	0.110	65	0.007
5	$Kd_{(av)} = 0.571 + 0.177\ g_{440} + 0.102\ Tn$	0.798	65	$\ll$ 0.001
6	$\ln Kd_{(av)} = 0.491\ \ln g_{440} + 0.203\ \ln Tn - 0.167$	0.852	65	$\ll$ 0.001
7	$Zeu = 0.224 + 1.265\ Z_{SD}$	0.881	51	$\ll$ 0.001
8	$\underline{b} = 0.376 + 1.186\ Tn$	0.887	23	$\ll$ 0.001

Table 3.4: Location and description of the sites surveyed in N.S.W. * denotes that the spectral distribution of P.A.R. was also measured for these waters. For additional limnological information, see Timms (1970, 1982) and Section 2.2. The sites can be located by number in Figure 3.2.

No.	Lake	Description and Use (plus additional references where appropriate)
1	Toonumbar Dam	Irrigation Water Supply, near Kyogle
2	Clarrie Hall Dam	Town Water Supply, Murwillumbah
3	Rocky Creek Dam	Town Water Supply, Lismore
4	*Lake Ainsworth	Coastal Dune Lake, Lennox Head (Bayly, 1964; Timms, 1982)
5	Cooks Lagoon	Coastal Dune Lake, Evans Head (Timms, 1982)
6	*Lake Minnie Water	Coastal Dune Lake, Wooli (Timms, 1969, 1982)
7	*Lake Hiawatha	Coastal Dune Lake, Wooli (Timms, 1969, 1982)
8	Karangi Dam	Off-creek storage reservoir. Town water supply, Coffs Harbour
9	Rosendahl Reservoir (Port Macquarie Dam)	Off-creek storage reservoir. Town water supply, Port Macquarie
10	Bootawa Dam	Off-creek storage reservoir. Town water supply, Taree
11	*Storm King Dam	Town Water Supply, Stanthorpe, Qld
12	Beehive Dam (Wallangarra Dam)	Town Water Supply, Wallangarra, Qld
13	St Helena Dam	Disused Water Supply for Meatworks, Wallangarra
14	Tenterfield Reservoir	Town Water Supply, Tenterfield
15	Ranger Valley Dam	Private Irrigation Dam, near Glen Innes
16	Beardy Waters Reservoir	Town Water Supply, Glen Innes
17	*Malpas Dam	Town Water Supply, Armidale
18	Puddledock Dam	Town Water Supply, Armidale
19	*Dumaresq Dam	Disused Town Water Supply. Now recreational use, Armidale
20	Gara Dam	Town Water Supply, Armidale
21	*Oaky Dam	Hydroelectric impoundment, east of Armidale
22	Kentucky Creek Dam	Town Water Supply, Uralla
23	Yalgoo Reservoir	Private Irrigation Supply, near Walcha
24	Glen Lyon Dam	Large Irrigation Storage, Mole River, Qld
25	*Pindari Dam	Large Irrigation Storage, near Ashford
26	Lake Inverell	Disused Town Water Supply, now a Wildlife Sanctuary, Inverell
27	*Copeton Dam	Large Irrigation Storage, near Inverell
28	*Lake Keepit	Large Irrigation Storage, near Gunnedah
29	Quipolly Dam	Town Water Supply, Werris Creek
30	*Chaffey Dam	Irrigation Storage and Town Water Supply, Tamworth (May & Powell, 1986)
31	Dungowan Dam	Town Water Supply, Tamworth
32	Glenbawn Dam	Town Water Supply and Irrigation Storage, Scone, Hunter Valley.
33	Lake Liddell	Supply of cooling water for Thermal Power Stations, near Muswellbrook, Hunter Valley
34	Lake St Clair (Glennies Creek Dam)	Town Water Supply and Irrigation Storage, Hunter Valley
35	Lostock Dam	Irrigation Water Storage, Hunter Valley
36	Chichester Dam	Town Water Supply, Hunter Valley
37	*Yarrie Lake	Turbid, shallow natural lake, near Narrabri

PART B NORTH-EAST NEW SOUTH WALES

3.3.7 Surface Temperature, pH, and Conductivity

Data for surface temperature, pH, and conductivity (at 18°C) are given in Table 3.5. Surface temperature varied depending on the time of year, location, and meteorological factors. The thermal stratification patterns of these waterbodies were not determined.

pH values ranged from 5.60 in Cooks Lagoon to 9.10 in Toonumbar Dam, with a median value of 8.05. These are generally higher than recorded previously for the region (e.g. Timms, 1969, 1982), but with the considerable enrichment with bicarbonates in many of these waters (Timms, 1970; Banens, in press), alkaline pH values are to be expected.

Conductivity (K_{18}) varied markedly between waterbodies, even between those in close proximity, so that no particular locational patterns were evident. The lowest measurement, 30.9 μs cm^{-1}, was for Oaky Dam, while the highest, 2140 μs cm^{-1}, was recorded in Lake Liddell. The median value was 150 μs cm^{-1}.

3.3.8 Total Nitrogen and Total Phosphorus

Data for these two parameters are also given in Table 3.5. Total nitrogen values ranged from 100 μg L^{-1} in Rosendahl Reservoir to 1460 μg L^{-1} in Yarrie Lake (median value 420 μg L^{-1}), while total phosphorus varied from 6 μg L^{-1} in Lake Minnie Water to 568 μg L^{-1} in Yarrie Lake. The median total phosphorus value for the area was 28 μg L^{-1}. Most of the dune lakes and off-creek storage reservoirs from along the coast, and the Hunter Valley reservoirs, had the lowest nutrient levels. The remaining waterbodies are eutrophic, and some are markedly so.

3.3.9 Turbidity, Colour and Chlorophyll *a*

The results for these parameters are listed in Table 3.5. Apart from Lake Ainsworth, the natural coastal lakes and off-creek storage reservoirs (Karangi, Rosendahl, Bootawa) were the least turbid waters of the area, along with the reservoirs of the Hunter Valley and some of the large impoundments of the Western Slopes. These, and two other reservoirs, Malpas and Gara Dams, all had turbidities of less than 1.5 N.T.U. The majority of the other waterbodies fall within the range 2.0 to 4.5 N.T.U. However a number exceeded this, including Quipolly and St. Helena Dams, where turbidity was due to blooms of

Table 3.5: Temperature, chemical, and optical data from freshwater lakes and reservoirs of north-east New South Wales

No.	Name	Date Sampled	Temperature and Chemical Data						Optical Data							
			Temp.	pH	K_{18}	Total N	Total P	Tn	g_{440}	Chlorophyll a	Secchi Disc	Kd (av)	Zeu	Ku (av)	$R_{(o)}$	b
			°C		μScm^{-1}	$\mu g\ L^{-1}$		N.T.U.	m^{-1}	$\mu g\ L^{-1}$	m	m^{-1}	m	m^{-1}	%	m^{-1}
1	Toonumbar Dam	26-11-86	27.0	9.10	147	820	25	3.5	0.461	14.31	1.35	1.23	3.65	1.06	3.65	5.37
2	Clarrie Hall Dam	15-7-87	15.4	7.40	83.4	600	-	6.2	2.648	5.73	1.40	1.66	2.66	1.56	5.04	6.31
3	Rocky Creek Dam	26-11-86	26.0	7.20	40.0	720	22	3.3	0.748	39.43	1.50	1.35	3.23	1.28	2.82	3.69
4	Lake Ainsworth	27-11-86	25.0	7.10	166	1240	78	2.1	16.351	41.89	0.90	3.89	0.92	4.22	0.38	2.61
5	Cooks Lagoon	27-11-86	26.0	5.60	75.0	330	8	0.45	7.197	1.92	>1.80	1.91	>1.70	-(A)	0.14	-(A)
6	Lake Minnie Water	25-8-86	18.0	8.30	104	160	6	0.92	0.921	4.24	3.45	0.84	5.12	0.81	0.94	0.86
7	Lake Hiawatha	25-8-86	17.0	8.30	104	280	11	0.50	0.058	2.01	>5.50	0.33	>5.00	-(B)	-(B)	-(B)
8	Karangi Dam	10-9-87	18.1	7.80	57.9	120	9	0.97	0.173	3.12	3.95	0.46	9.50	0.47	2.37	1.30
9	Rosendahl Reservoir	11-9-87	17.0	7.60	132	100	7	1.2	0.115	3.50	2.45	0.55	8.52	0.51	3.02	1.53
10	Bootawa Dam	11-9-87	17.2	7.60	136	110	16	1.8	0.748	2.54	3.75	0.65	6.71	0.63	3.77	2.06
11	Storm King Dam	11-12-86	23.4	8.35	99.6	600	34	2.4	2.706	10.18	1.80	1.49	3.40	1.46	2.00	2.37
12	Beehive Dam	14-7-87	9.8	8.20	31.4	350	20	2.5	2.015	5.84	1.90	1.43	2.89	1.39	1.68	2.76
13	St. Helena Dam	11-12-86	25.0	8.85	57.9	1060	86	20.0	11.227	58.36	0.50	3.92	0.95	4.12	5.60	21.94
14	Tenterfield Res.	12-12-86	22.2	8.20	205	710	26	3.5	1.555	1.51	2.40	1.19	3.76	1.05	3.98	3.26
15	Ranger Valley Dam	12-12-86	21.8	8.75	447	890	146	2.2	0.921	15.63	2.65	0.95	4.39	0.87	1.75	2.03
16	Beardy Waters Res.	26-7-87	7.5	8.20	377	560	88	5.7	2.821	20.88	1.15	1.58	2.65	1.50	5.92	5.96
17	Malpas Dam	4-11-86	18.2	8.60	233	410	30	1.3	1.267	8.37	3.05	0.81	5.70	0.68	1.32	1.48
18	Puddledock Dam	13-8-86	10.0	8.20	509	1090	62	4.4	1.612	44.26	0.95	1.72	2.68	1.66	3.40	5.93
19	Dumaresq Dam	31-7-86	10.0	8.35	150	830	54	3.6	6.506	22.23	1.20	1.68	2.41	1.58	2.34	4.51
20	Gara Dam	13-8-86	11.0	7.95	482	300	31	1.4	0.461	7.86	2.45	0.91	4.69	0.72	2.20	2.47
21	Oaky Dam	4-9-86	12.9	8.15	30.9	420	42	4.4	4.606	17.06	1.75	1.59	2.53	1.54	2.21	3.39
22	Kentucky Creek Dam	7-1-87	27.0	7.95	85.7	570	159	47.0	12.494	13.82	0.50	3.94	1.05	3.56	13.98	38.59
23	Yalgoo Reservoir	7-1-87	23.0	8.00	125	1070	133	3.6	5.470	15.15	1.40	1.72	2.46	1.52	1.70	3.93
24	Glen Lyon Dam	14-7-87	14.0	7.45	239	550	19	1.2	0.058	8.21	2.40	1.23	3.70	-(C)	2.75	-(C)
25	Pindari Dam	26-7-87	11.9	7.85	250	270	22	1.3	1.036	1.93	4.00	0.62	6.97	0.57	2.00	1.52
26	LakeInverell	25-7-87	10.7	7.95	685	330	123	7.4	1.497	23.10	0.75	1.97	2.35	1.76	6.75	9.33
27	Copeton Dam	25-7-87	12.7	7.80	180	350	31	1.7	1.036	7.22	3.50	0.74	5.96	0.72	3.29	2.21
28	Lake Keepit	19-12-86	24.1	8.10	228	400	34	4.0	0.748	9.22	1.50	0.94	4.74	0.81	4.38	3.14
29	Quipolly Dam	19-7-87	11.1	8.65	521	1200	174	6.8	0.345	286.66	0.80	2.48	1.70	2.29	4.15	10.92
30	Chaffey Dam	21-10-86	20.6	8.10	329	460	38	2.7	1.036	6.15	2.60	0.95	4.64	0.87	2.71	2.69
31	Dungowan Dam	21-10-86	16.3	8.10	52.2	470	20	2.5	0.979	5.62	2.15	0.84	5.43	0.70	3.35	2.40
32	Glenbawn Dam	19-7-87	12.8	7.90	332	310	15	0.81	0.173	8.72	3.35	0.68	6.30	0.65	2.93	1.69
33	Lake Liddell	18-7-87	15.2	8.05	2140	350	23	1.0	0.058	12.61	3.10	0.63	6.80	0.77	3.35	1.68
34	Lake St. Clair	18-7-87	13.0	7.55	234	280	11	0.73	0.576	6.36	3.10	0.85	4.97	0.84	1.65	1.46
35	Lostock Dam	17-7-87	12.2	7.85	199	110	11	1.2	0.806	2.65	3.60	0.58	7.38	0.57	2.65	1.91
36	Chichester Dam	17-7-87	12.8	7.50	65.5	190	14	1.1	0.633	15.05	3.60	0.70	6.27	0.70	1.99	1.63
37	Yarrie Lake	18-12-86	26.1	8.05	146	1460	568	425	11.515	2.42	0.015	48.20	0.08	36.5	11.40(D)	283.3
	MEDIAN		17.0	8.05	150	420	28	2.4	1.036	8.37	2.15	1.19	4.64	1.05	2.79	2.65
	MEAN (E)		17.1	7.96	256	517	47	4.3	2.495	20.93	2.20	1.36	4.33	1.32	3.20	4.90
	STANDARD DEVIATION (E)	±	5.8	0.60	361	325	47	8.1	3.643	47.47	1.08	0.93	2.17	0.96	2.38	7.24

(A) Too shallow for calculation
(B) Upwelling irradiance not measured
(C) Instrument failure during profile
(D) $R_{(o)}$ is for 0.05 metre depth
(E) Excluding Yarrie Lake

photosynthetic organisms, and Clarrie Hall and Kentucky Creek Dams, which had high tripton loadings. The greatest turbidity recorded, 425 N.T.U., was for the natural, shallow Yarrie Lake. The median turbidity for the lakes studied was 2.40 N.T.U.

Most of the waterbodies investigated had low to moderate levels of dissolved organic colour, and only twelve had gilvin values exceeding 2.0 m^{-1}. The more turbid waters also tended to be the more dystrophic. St. Helena and Kentucky Creek Dams, and Yarrie Lake, all had high levels of gilvin. The coastal dune lakes were extremely variable in humic content, with Lake Ainsworth being the most dystrophic of all sites studied, and Lake Hiawatha one of the least. The median g_{440} value, 1.036 m^{-1}, reveals how uncoloured most of the study sites were.

The chlorophyll a content of the waters was highly variable, being as low as 1.51 $\mu g\ L^{-1}$ in Tenterfield Reservoir, and reaching a maximum of 287 $\mu g\ L^{-1}$ in Quipolly Dam, with the median value being 8.37 $\mu g\ L^{-1}$. Chlorophyll a concentrations exceeded 10 $\mu g\ L^{-1}$ in sixteen of the locations.

3.3.10 The Attenuation of P.A.R.

Values for the mean downwelling vertical attenuation coefficient ($Kd_{(av)}$), and the measured euphotic depth (Zeu), are given in Table 3.5. These are single measurements only, and do not take account of seasonal variations, which may be considerable (Scribner, in Kirk, 1986). Profiles of light penetration with depth are given for selected lakes with different optical properties in Figure 3.10. Lake Hiawatha ($Kd_{(av)}$ = 0.33 m^{-1}) was the clearest, but as gilvin, turbidity, and chlorophyll a concentrations increased, so did attenuation with a consequent decrease in euphotic depth. However, most lakes and reservoirs had $Kd_{(av)}$ values of less than 2.0 m^{-1}, and eighteen had values below 1.0 m^{-1}. Attenuation was more rapid in a few of the turbid, highly coloured, and eutrophic waters, such as St. Helena Dam ($Kd_{(av)}$ = 3.92 m^{-1}), and the greatest attenuation occurred in highly turbid Yarrie Lake, where $Kd_{(av)}$ was an extremely high 48.2 m^{-1}, and 99% of incident light had been attenuated eight centimetres below the surface. The typical biphasic curve, with

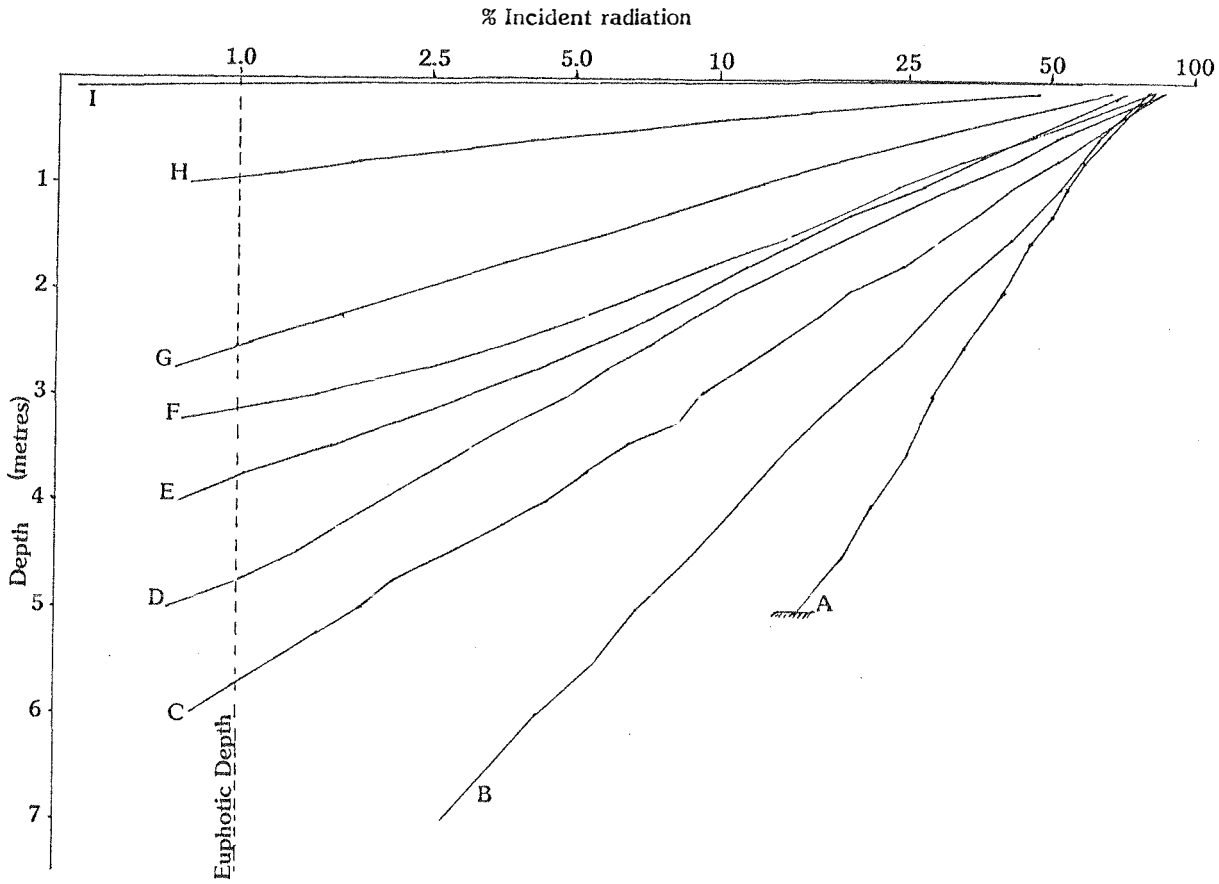


Figure 3.10 Attenuation profiles of downwelling P.A.R. (400-700 nm.) for selected north-east New South Wales lentic freshwaters. A. Lake Hiawatha, 25-08-1986; B. Karangi Dam, 10-09-1987; C. Malpas Dam, 04-11-1986; D. Lake Keepit, 19-12-1986; E. Tenterfield Reservoir, 12-12-1986; F. Storm King Dam, 11-12-1986; G. Oaky Dam, 04-09-1986; H. St. Helena Dam, 11-12-1986; I. Yarrie Lake, 18-12-1986. See Table 3.5 for further data on the optical properties of these waters.

attenuation greatest close to the surface, was apparent in many of the lakes (Figure 3.10). A marked increase in attenuation also occurred below about 2.50 metres in Storm King Dam, probably due to zonation of the phyto-flagellate *Ceratium hirundenella* (Müller) Schrank, which was present at those depths, but absent from the surface waters. A similar, smaller change was apparent in Tenterfield Reservoir. The median vertical attenuation coefficient, $Kd_{(av)}$, for the area was 1.19 m^{-1}, while that for euphotic depth was 4.64 metres.

Mean upwelling vertical attenuation coefficients ($Ku_{(av)}$) were generally not greatly different from the corresponding $Kd_{(av)}$ for each lake, although they were usually slightly lower. Any differences were probably due to decreased accuracy in measuring the much lower intensities of upwelling irradiance, especially at depth. Most locations were sufficiently deep to avoid any contribution from light reflected off the bottom to upwelling irradiance.

3.3.11 Reflectance and the Scattering Coefficient

Reflectance close to the surface, at 0.10 metres (Ro), is given in Table 3.5. This usually exceeded 1.5% and was highest in the more turbid locations, but varied considerably when turbidity was lower. Changes in reflectance with depth are shown for a number of waters of differing turbidity in Figure 3.11. Reflectance initially increased with depth, and in most instances continued to do so. This was especially marked in Kentucky Creek Dam and in Lake Inverell. In some, notably Malpas and Puddledock Dams, Lake Minnie Water, and Rosendahl Reservoir, reflectance tended to approach an asymptotic value after the initial increase, but in St. Helena and Storm King Dams decreased after peaking at shallower depths.

Values for the scattering coefficient, $\underline{b}$, ranged from 283 m^{-1} in Yarrie Lake, to 0.86 m^{-1} in Lake Minnie Water. The median value was 2.65 m^{-1}.

3.3.12 The Spectral Distribution of Underwater P.A.R.

The spectral distribution of P.A.R. (400-740 nm) was measured in twelve of the lakes and reservoirs. These are shown in Figure 3.12(a-c).

The distribution of underwater light in the clearest lake, (Lake Hiawatha), was centred at 580 nm, with almost equal extinction of the wavelengths either side of this, specifying a predominantly green to yellow underwater light climate. Attenuation of blue light increased

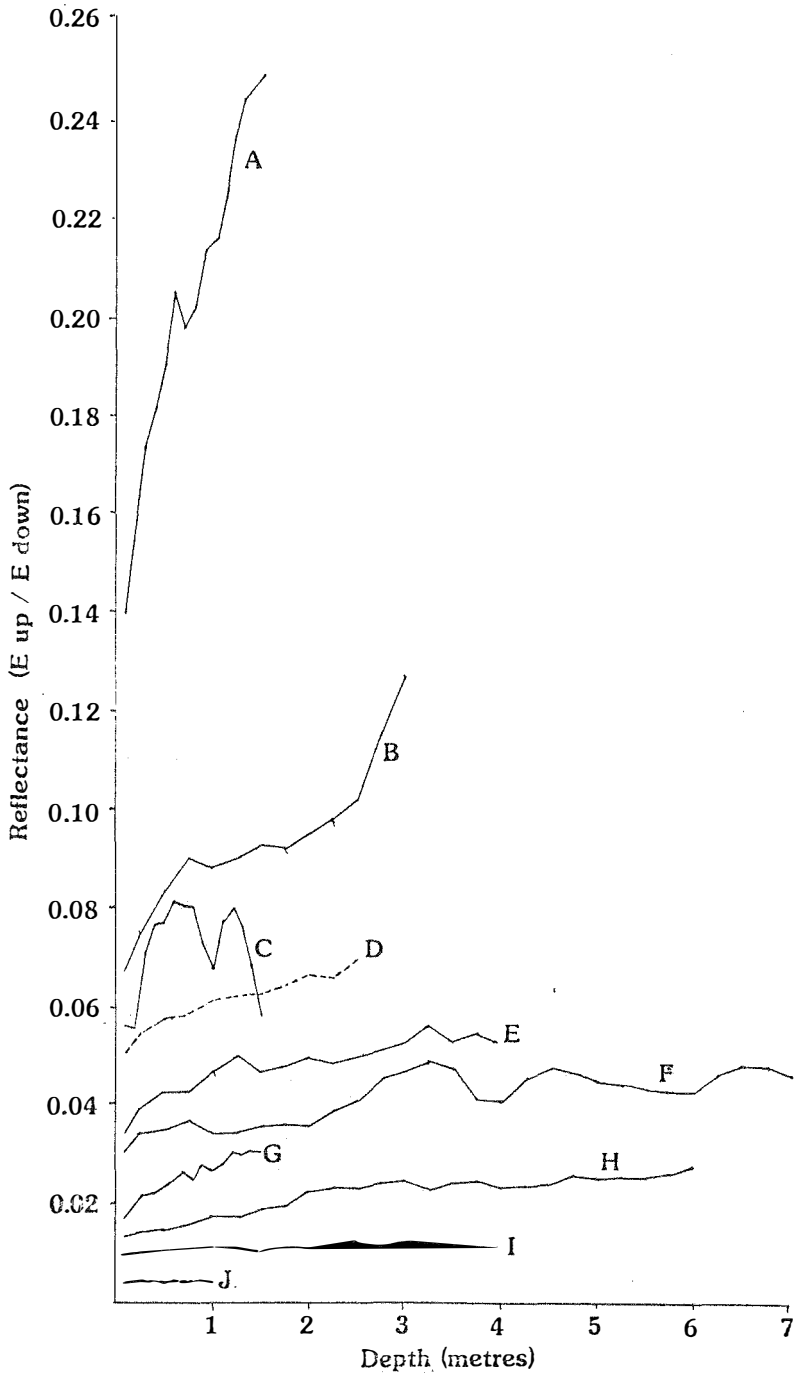


Figure 3.11 Reflectance/depth profiles for selected north-east New South Wales lentic freshwaters. A. Kentucky Creek Dam. 07-01-1987; B. Lake Inverell, 25-07-1987; C. St. Helena Dam, 11-12-1986; D. Clarrie Hall Dam (dashed line), 15-07-1987; E. Puddledock Dam, 13-08-1986; F. Rosendahl Reservoir, 11-09-1987; G. Yalgoo Reservoir, 07-01-1987; H. Malpas Dam, 04-11-1986; I. Lake Minnie Water, 26-08-1986; J. Lake Ainsworth, 27-11-1986.

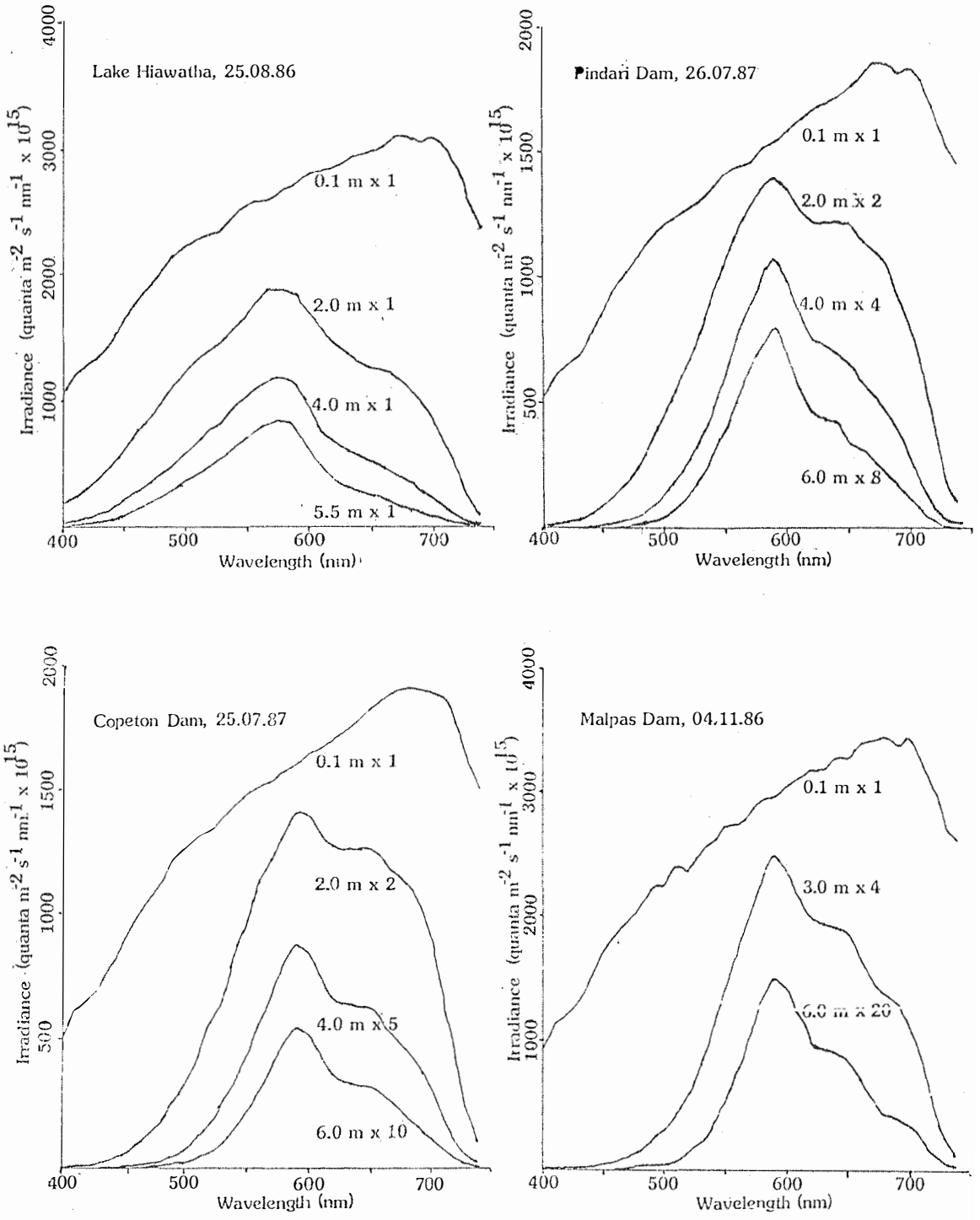


Figure 3.12(a) Spectral distributions of P.A.R. from north-east New South Wales lentic freshwaters - the clearer water lakes and reservoirs.

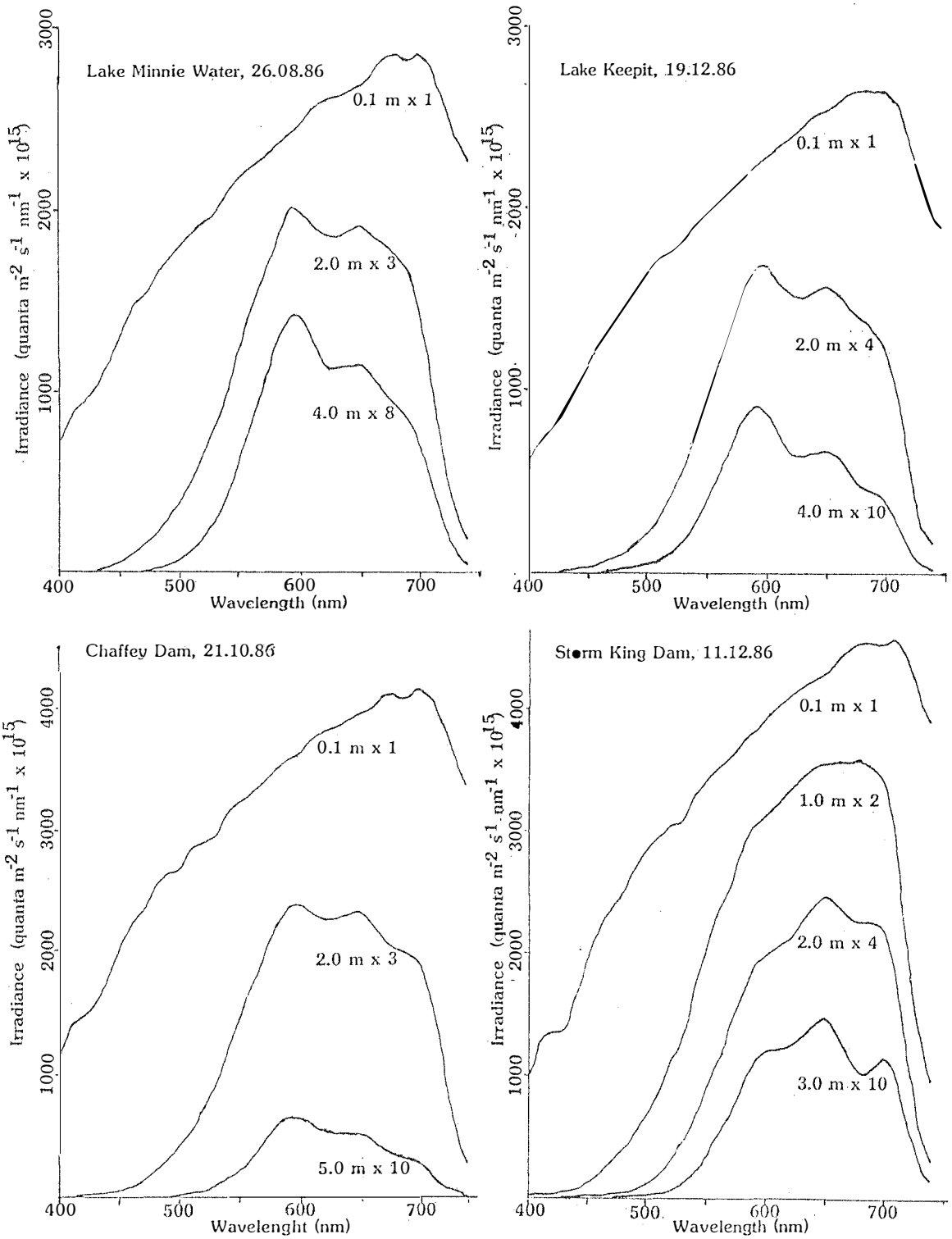


Figure 3.12(b) Spectral distributions of P.A.R. from north-east New South Wales lentic freshwaters - slightly coloured or turbid lakes and reservoirs.

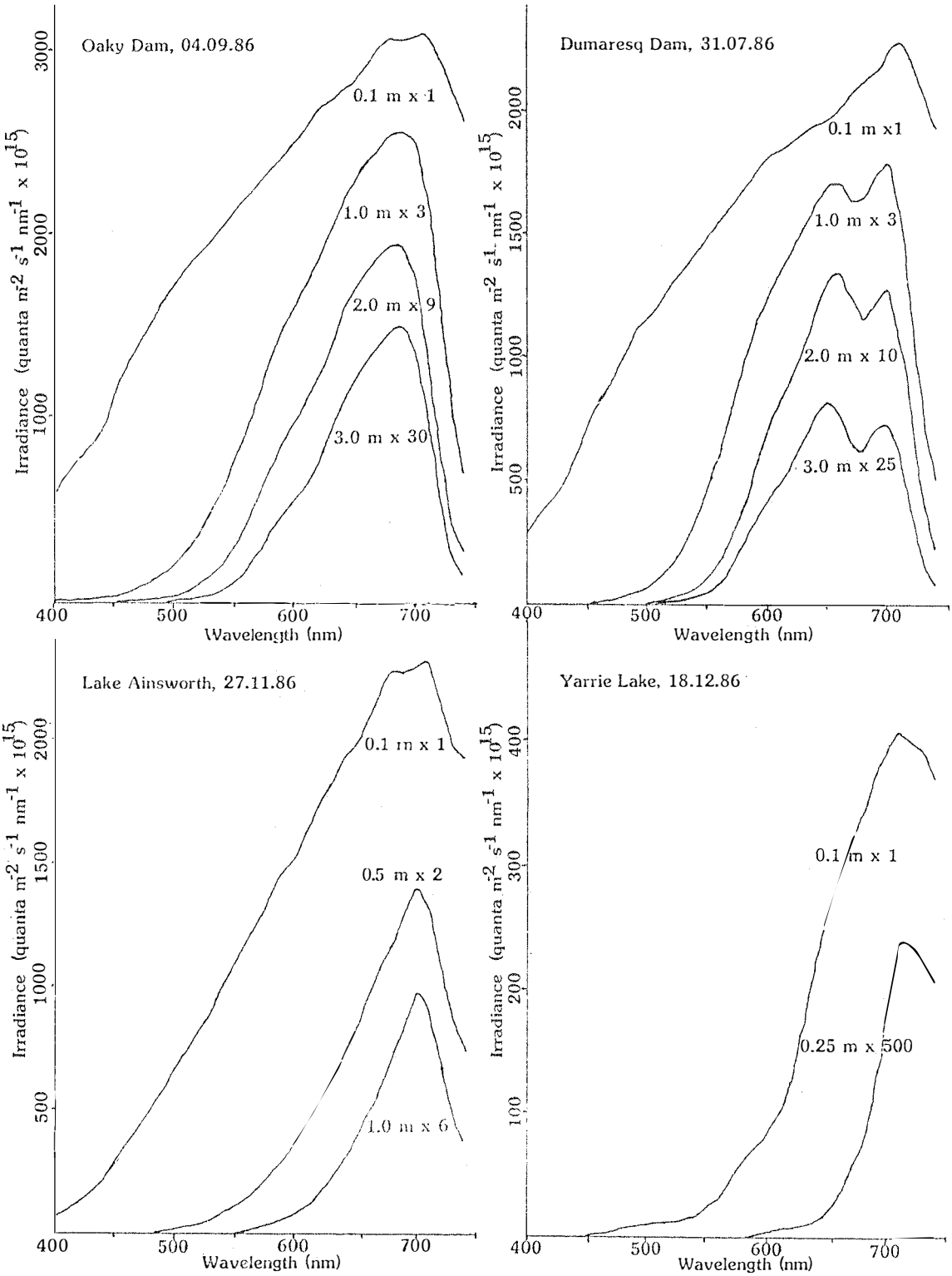


Figure 3.12(c) Spectral distributions of P.A.R. from north-east New South Wales lentic freshwaters - the more dystrophic or turbid lakes and reservoirs.

in comparison to red light as concentrations of gilvin and tripton increase. This is shown in the series Pindari, Copeton, and Malpas Dams, and Lake Minnie Water, where wavelengths shorter than 500 nm are extinguished progressively more rapidly within the first few metres below the surface. Despite this, these waters are still clear enough so that the wavelength of peak transmission changed only slightly, to 590-600 nm. The slightly turbid but low coloured waters of Chaffey Dam and Lake Keepit also quickly extinguish blue light, but even here the wavelength of maximum transmission remained near 600 nm. However, the spectral distributions of these reservoirs display a broad plateau of wavelengths between 590 and 660 nm, rather than a single peak wavelength. This is interupted by a shallow absorption trough centred on 620 nm. This was also present in the underwater spectrum of Lake Minnie Water, and as a shoulder on scans from Pindari, Copeton, Malpas, and Storm King Dams. The cause of this absorption trough cannot be explained (see Section 3.4.1, below).

The spectral distribution of P.A.R. in Storm King Dam showed changes due to this reservoir's higher gilvin content, with a broad transmission peak now centred at 660 nm. The probable stratification of *Ceratium hirundinella* caused a marked absorption trough centred at 680 nm in the spectral distribution of light below two metres, corresponding to absorption by chlorophylla. A bloom of the diatom *Asterionella formosa* Hass. also influenced the spectral distribution of underwater P.A.R. in Dumaresq Dam, causing a similar absorption trough at 680 nm. Considerable gilvin concentrations, as well as phytoplankton and tripton, all contribute to the rapid attenuation of the shorter wavelengths in this reservoir.

The spectral scans from Oaky Dam and Lake Ainsworth are typical of humic waters. Maximum transmission in moderably humic Oaky Dam was at 680 to 690 nm, and at 700 nm in highly humic Lake Ainsworth, where the transmission window had also been further narrowed. Any contribution from the high chlorophyll *a* contents of these waters is not obvious.

The underwater spectrum in extremely turbid Yarrie Lake was very similar to those from humic lakes, but the transmission peak was even further to the right, being centred on 710 nm. Most wavelengths below 650 nm were extinguished within the surface twenty-five centimetres of water.

3.3.13 Secchi Disc Transparency

Measurements of Secchi disc transparency are given in Table 3.5. Secchi disc depth exceeded 3.0 metres in twelve of the study sites, being greatest in Lake Hiawatha, but were shallower than 1.0 metre in seven, including polyhumic Lake Ainsworth, the turbid Lake Inverell, St. Helena and Kentucky Creek Dams, and in Quipolly and Puddledock Dams, where blooms of phytoplankton were present. The minimum value, 0.015 metres, was in Yarrie Lake. The medium value was 2.15 metres.

3.3.14 Regression and Correlation Analyses

Pearson correlation analysis was conducted between most of the parameters listed in Table 3.5, and linear and multiple regression analyses undertaken using either $Kd_{(av)}$ or b as the dependent variable, and turbidity, g_{440}, and chlorophylla as the independent variables. Data from Yarrie Lake were excluded, as these were considered atypical of the study area, and may have placed considerable bias on the results of these analyses.

Highly significant correlations were found between most optical properties. These included positive correlations between $Kd_{(av)}$ and turbidity, g_{440}, and chlorophylla (see Table 3.6), and negative correlations between Secchi disc depth and all other optical parameters. Additionally, g_{440} and turbidity were positively correlated ($r = 0.499$, $P = 0.002$), but chlorophylla was not significantly correlated with either. Thus, most of the suspended particulate matter may have been mineral particle, detritus, or other tripton, rather than phytoplankton.

Several other significant correlations are of note. Water colour and temperature were weakly positively correlated ($r = 0.4154$, $P = 0.012$), possible because coloured waters rapidly absorb most solar radiation at shallow depth, therefore causing warmer surface temperatures. This hypothesis is supported by a further but weak significant positive correlation between temperature and $Kd_{(av)}$ ($r = 0.3545$, $P = 0.032$). The rapid attenuation of light in the surface waters of humic lakes, resulting in their warming, may possible contribute to such a relationship. However, these correlations may also be artifacts of seasonal sampling, as the most highly coloured lakes ($g_{440} > 10.0\ m^{-1}$) were all sampled during summer. Seasonal data are unavailable to test these relationships for other times of the year. Both nutrients were significantly correlated with chlorophylla (Total nitrogen: chlorophylla, $r = 0.5365$, $P << 0.001$; Total phosphorus: chlorophylla, $r = 0.5635$, $P << 0.001$), revealing algal responses to increased nutrient loadings. Total nitrogen was also significantly

Table 3.6: Regression analysis showing the relationships between various optical characteristics of north-east New South Wales standing freshwaters.

Equation No.	r^2	n	P
1 $Kd_{(av)} = 0.812 + 0.220\ g_{440}$	.750	36	$\ll 0.001$
2 $Kd_{(av)} = 1.02 + 0.0795\ Tn$	.481	36	$\ll 0.001$
3 $Kd_{(av)} = 1.20 + 0.0075\ Chla$	.148	36	$0.20 < P < 0.05$
4 $Kd_{(av)} = 0.75 + 0.04\ Tn + 0.176\ g440$	.841	36	$\ll 0.001$
5 $Kd_{(av)} = 0.689 + 0.215\ g_{440} + 0.007\ Chla$	.861	36	$\ll 0.001$
6 $Kd_{(av)} = 0.916 + 0.075\ Tn + 0.006\ Chla$	.561	36	$\ll 0.001$
7 $Kd_{(av)} = 0.646 + 0.035Tn + 0.176g_{440} + 0.006Chla$	.931	36	$\ll 0.001$
8 $Zeu = 0.386 + 1.78\ Z_{SD}$	.796	35	$\ll 0.001$
9 $\underline{b} = 0.948 + 0.85\ Tn$	.965	33	$\ll 0.001$

correlated with colour ($r = 0.5174$, $P = 0.002$), probably resulting from nitrogen forming part of the elemental composition of dissolved humus, as was total phosphorus with turbidity ($r = 0.5683$, $P << 0.001$). This could be due mainly to phosphorus being absorbed onto the outer surface of tripton particles.

Despite wide variation in temperature due to seasonal and locational differences, no significant correlation existed between temperature and chlorophyll a, suggesting that water temperature had little influence on the quantity of phytoplankton present. Also, no significant correlation was found between pH and the humic content of the waters, possibly because of buffering by the dominant anion, bicarbonate (Timms, 1970; Banens, in press).

The results of regression analyses are shown in Table 3.6. Natural logarithmic transformations did little to improve the relationships between the parameters of these regressions, and are therefore omitted. Dissolved organic colour is the major attenuator of light in these waters, although turbidity is also a significant contributor, and chlorophyll a a minor one. All three components account for almost all the variability observed in $Kd_{(av)}$, and turbidity is the major factor influencing the scattering coefficient, b.

The relationship between Secchi disc depth and euphotic depth was also examined, and a highly significant positive correlation found between these.

3.3.15 Phytoplankton

The dominant phytoplankton found in the single samples from each study site are given in Table 3.7. These data give an indication of the considerable range of phytoplankton present in north-east New South Wales freshwaters, despite their lack of seasonality and that some taxa have been identified only to genus level. Additionally some of the smaller species may well have been missed by the sampling method, passing through the 20 µm mesh of the plankton net.

Algae from the Division Chlorophyceae were particularly common, especially desmids and colonial species such as *Botryococcus braunii* Kützing, *Dictyosphaerium* spp., *Oocystis* spp. and a coccoid alga of the *Sphaerocystis schroeteri* Chodat type. *Staurastrum* was the main desmid genus present, and frequently encountered species included *S. freemanii* West & West, *S. sonthalianum* Turner, *S. paradorum* Meyen, *S. sexangulare* (Bulnheim) Rabenhorst, and *S. tohopekaligenese* Wolle, while a triradiate

| Lake No. | LAKE / ALGA | | |

species with affinities to *S. pingue* Teiling was present in most reservoirs of the New England Tablelands. Flagellated green algae such as *Volvox* sp. and *Eudorina elegans* Ehrenberg were also prominent components of the phytoplankton at several locations.

Dinoflagellates, in particular *Peridinum* spp. (especially *P. volzii* Lemmerman), and *Ceratium hirundinella* (Müller) Schrank were also abundant in many locations. *C. hirundinella* formed an almost monospecific bloom in Quipolly Dam. *Prorocentrum palyfairi* Croome & Tyler was recorded from three coastal locations, Toonumbar Dam, Cooks Lagoon, and Lake Hiawatha. Euglenoids were also present in most samples, with *Trachelomonas volvacina* Ehrenberg, *T. hispida* (Perty) Stein, and *T. armarta* (Ehrenberg) Stein being the most common, while *Euglena oxyuris* Schmarda was the dominant alga in Cooks Lagoon. Chrysophytes were much less common, and when present were usually *Dinobryon cylindricum* Imhoff or *D. sertularia* Ehrenberg. A long-spined species of *Mallomonas* formed an almost monospecific bloom in Puddledock Dam and *M. splendens* (G.S. West) Playfair was recorded from Lake Hiawatha, and Glenbawn, Beehive, Clarrie Hall, Chichester and Storm King Dams.

Diatoms were locally abundant. *Melosira granulata* (Ehrenberg) Ralfs and *Rhizosolenia eriensis* H.L. Smith were present in a number of the sites, while *Asterionella formosa* Hass. formed a dense bloom in Dumaresq Dam. *Anabaena flos-aquae* (?) (Lyngbye) Brebisson and *Microcystis aeruginosa* Kützing emend. Elenkin were the main blue-green algae present.

PART C SOUTH-EAST QUEENSLAND

3.3.16 Temperature, pH, and Conductivity at 18°C (K_{18})

Surface temperatures, pH, and K_{18} for each lake are presented in Table 3.8. Temperatures ranged from 21.0°C to 25.2°C, probably resulting from day by day variations in weather, during the three week sampling period, rather than underlying differences between the lakes themselves. Most of the lakes are isothermal and polymictic (Bayly et al, 1975; Arthington et al, 1986).

Red Lagoon had the lowest pH, 4.25, whilst the highest, 7.55, was from Lake Minker. pH values were higher than previously reported in all but Brown Lake, and four were above neutral. The median value was 5.60. Secchi disc depths indicate that the humic content of many lakes was lower during this survey than in previous studies, and with a strong positive correlation existing between water transparency and pH (Bayly et al, 1975), with greater water clarity, higher pH values are also likely.

The K_{18} measurements ranged from 52.1 μS cm^{-1} to 576 μS cm^{-1}, with the median value being 96.7 μS cm^{-1}. These are close to those previously reported for the lakes (Bayly, 1964; Bayly et al, 1975; Miller et al, 1984; Little and Roberts, 1983; Reeve et al, 1985; Bensink, 1976; Bensink and Burton, 1975; Arthington and Watson, 1982), except for Lakes Wabby, Woonjeel, Minker, and Ocean Lake, which were considerably higher. This may result from evaporative concentration of ions in these lakes, where considerable amounts of exposed beach indicated lower than normal water levels. These four lakes also had pH values above 7.0, possibly due to their greater ionic concentrations allowing greater buffering capacities (see also Chapter Four).

3.3.17 Total Nitrogen and Total Phosphorus

Nutrient levels were generally low, and most lakes were oligotrophic. Values are given in Table 3.8. Total nitrogen values ranged from 1060 μg L^{-1} in the southernmost of the Boomerang Lakes, to below the level of detection in several, including Lakes Freshwater, McKenzie and Birrabeen, and in Honeyeater Lake (Median value = 150 μg L^{-1}). Twenty-one of the lakes had total phosphorus levels of 10 μg L^{-1} or less, with a minimum value of 3 μg L^{-1} in Lake Birrabeen on Fraser Island, Blue Lagoon on Moreton Island, and Blue Lake on North Stradbroke Island. The maximum amount of total phosphorus recorded in these lakes was 20 μg L^{-1}, in Ocean Lake.

Table 3.8: Temperature, chemical, and optical data for the freshwater coastal dune lakes of south-east Queensland. Locations of the lakes are shown in Figure 3.3. A - indicates that no measurements were made.
ND = Below the level of detection.

No.	Name	Temperature and Chemical Data						Optical Data							
		Temp.	pH	K_{18}	Total N	Total P	Tn	g_{440}	Chlorophyll a	Secchi Disc	$Kd_{(av)}$	Zeu	$Ku_{(av)}$	$R_{(o)}$	b
		°C		$\mu S\ cm^{-1}$	$\mu g\ L^{-1}$	$\mu g\ L^{-1}$	N.T.U.	m^{-1}	$\mu g\ L^{-1}$	m	m^{-1}	m	m^{-1}	%	m^{-1}
	FRASER ISLAND														
1	Lake Minker	24.2	7.55	576	400	9	0.72	1.094	1.90	> 2.80	0.71	-	-	-	-
2	Lake Woonjeel	24.7	7.30	507	400	8	0.44	1.036	2.64	> 2.30	0.84	-	-	-	-
3	Ocean Lake	23.8	7.40	243	180	20	0.91	1.497	12.82	2.45	1.09	> 3.25	0.98	1.06	1.37
4	White Lake	23.6	5.20	75.4	260	17	1.60	1.267	7.99	1.80	0.92	> 2.75	0.90	2.41	2.30
5	Lake Bowarrady	24.0	6.45	82.6	320	7	0.63	0.288	6.18	> 2.50	0.58	-	-	-	-
6	Lake Allom	23.9	5.40	58.4	170	10	0.88	0.691	5.47	3.45	0.71	> 3.50	-	-	-
7	Deepwater Lake	23.5	6.25	99.3	20	4	0.53	0.000	2.43	> 4.55	0.32	-	-	-	-
8	Lake Coomboo	24.2	5.15	63.2	320	10	1.20	7.715	11.77	1.40	1.96	2.00	1.72	0.39	2.08
9	Hidden Lake	23.6	5.10	57.0	130	10	0.58	3.224	1.80	2.90	1.23	3.38	1.24	0.29	0.58
10	Boomerang Lake (north)	24.5	5.20	98.1	120	9	1.30	0.864	2.57	> 1.65	1.33	-	-	-	-
11	Boomerang Lake (south)	25.2	5.10	116	1060	6	1.00	9.385	1.37	> 0.75	3.47	-	-	-	-
12	Lake Garawongera	21.7	5.10	67.7	160	6	0.59	1.209	3.07	> 4.25	0.81	-	-	-	-
13	Lake Wabby	24.1	7.40	213	110	9	1.00	0.058	4.59	2.85	0.48	> 7.00	0.37	3.08	1.53
14	Lake McKenzie	23.3	5.80	95.3	ND	5	0.27	0.058	0.21	> 7.75	0.19	-	-	-	-
15	Basin Lake	24.8	6.40	52.1	400	6	1.00	0.461	1.69	5.65	0.49	> 6.50	0.42	1.18	0.63
16	Jennings Lake	24.0	5.05	87.0	90	6	0.55	4.145	-	> 2.90	1.30	-	-	-	-
17	Lake Birrabeen	23.0	5.60	72.9	ND	3	0.40	0.403	0.53	> 5.60	0.26	-	-	-	-
18	Lake Benaroon	22.0	5.45	88.2	120	4	0.81	0.979	3.28	> 2.60	0.91	-	-	-	-
19	Lake Boomanjin	21.2	5.90	110	40	9	0.53	2.591	3.17	2.90	1.13	3.71	1.23	0.51	1.14
20	Red Lagoon	21.2	4.25	115	480	12	1.00	27.866	1.06	0.65	4.91	0.73	-	-	-
	COOLOOLA SAND MASS														
21	Lake Freshwater	21.0	6.40	135	ND	5	1.00	0.000	11.02	> 1.00	0.66	-	-	-	-
22	Lake Cooloomera	22.0	5.15	162	440	17	0.55	14.221	6.04	1.30	3.15	1.21	-	-	-
	MORETON ISLAND														
23	Blue Lagoon	22.0	5.60	110	4	3	0.51	0.000	4.56	> 4.20	0.39	> 3.75	-	-	-
24	Honeyeater Lake	22.4	5.90	112	ND	9	0.81	0.000	6.63	3.60	0.40	> 4.25	-	-	-
	NORTH STRADBROKE ISLAND														
25	Brown Lake	23.0	5.05	61.3	220	19	3.00	1.958	15.89	0.85	2.88	1.42	2.53	2.07	6.32
26	Blue Lake	23.4	6.40	67.0	ND	3	0.62	0.086	3.49	6.35	0.42	> 6.50	0.32	1.04	0.57
	MEDIAN	23.5	5.60	96.7	150	9	0.77	1.008	3.28	-	0.83	-	-	1.18	1.37
	MEAN	23.2	5.83	136	211	9	0.86	3.119	4.89	-	1.21	-	-	1.34	1.84
	STANDARD DEVIATION	1.2	0.87	128	232	4.6	0.53	6.072	4.12	-	1.16	-	-	0.97	1.80

3.3.18 Turbidity, Colour and Chlorophylla

Turbidity, colour (as g_{440}), and chlorophyll a measurements are also listed in Table 3.8. Turbidity was low in all lakes, and only four exceeded 1.0 N.T.U. Brown Lake, North Stradbroke Island, was the most turbid, at 3.0 N.T.U., followed by White Lake on Fraser Island, at 1.60 N.T.U. Median turbidity for all twenty-six lakes was 0.77 N.T.U.

Dissolved humic substances were below the level of detection in several of the lakes, and most were only slightly to moderately coloured. Only seven had g_{440} values greater than 2.0 m^{-1}, the two highest being 27.866 m^{-1} in polyhumic Red Lagoon, and 14.221 m^{-1} in Lake Cooloomera. The median g_{440} value was 1.008 m^{-1}.

The highest chlorophyll a concentration, 15.9 $\mu g\ L^{-1}$, was in Brown Lake, and planktonic algae may have considerably enhanced the turbidity of the water there. Chlorophyll a also exceeded 10 $\mu g\ L^{-1}$ in Lakes Coomboo and Freshwater, and in Ocean Lake, whilst the lowest values were recorded for clear-water Lakes McKenzie and Birrabeen. The lakes had a median chlorophyll a value of 3.29 $\mu g\ L^{-1}$. The sample for Jennings Lake was misplaced between filtration and analysis.

3.3.19 The Attenuation, Reflectance and Scattering of P.A.R.

Attenuation was measured from an inflatable rubber dinghy anchored near the centre of each lake. This may not necessarily correspond with the deepest point in each lake.

Values of both the mean downwelling and upwelling vertical attenuation coefficients, $Kd_{(av)}$ and $Ku_{(av)}$, and the scattering coefficient $\underline{b}$ are presented in Table 3.8. $Kd_{(av)}$ values range from 0.19 m^{-1} in Lake McKenzie, the clearest lake, to 4.91 m^{-1} in highly dystrophic Red Lagoon, the median value being 0.83 m^{-1}. Gilvin was the main component causing increased attenuation coefficients, although phytoplankton may have contributed in Brown and Ocean Lakes, and turbidity in White and Brown Lakes. The influence of these attenuating components are illustrated for some lakes in Figure 3.13. Biphasic curves are apparent in most, the change in slope occurring below about 2.0 metres in the clearer water lakes. In dystrophic waters, such as Lake Cooloomera and Red Lagoon, the change in slope is much shallower, at less than 0.5 metres, resulting from the rapid extinction of blue light in these waters by gilvin. Weather

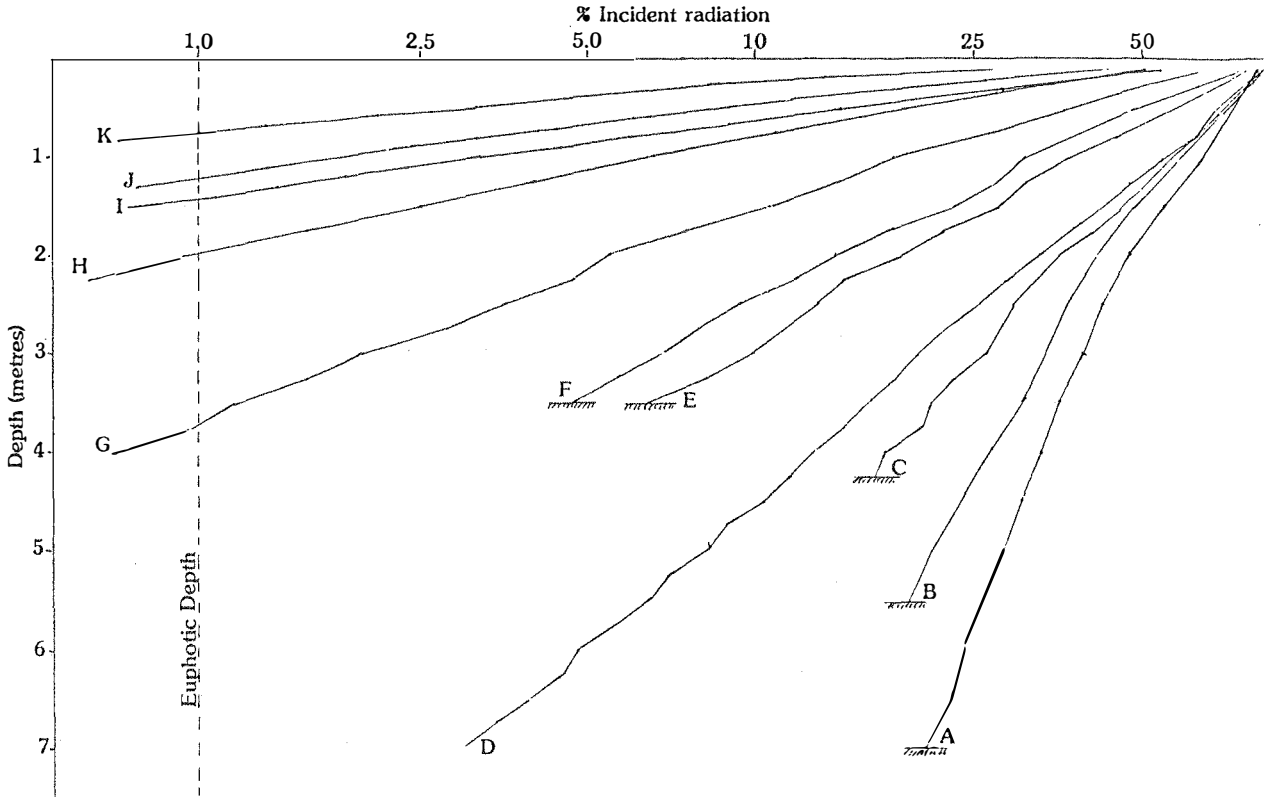


Figure 3.13 Attenuation profiles of downwelling P.A.R. (400-700 nm.) for selected south-east Queensland coastal lakes, April/May, 1987. A. Lake McKenzie; B. Lake Birrabeen; C. Honeyeater Lake; D. Lake Wabby; E. Lake Allom; F. Lake Garawongera; G. Lake Boomanjin; H. Lake Coomboo; I. Brown Lake; J. Lake Cooloomera; K. Red Lagoon. See Table 3.8 for further data on the optical characteristics of these lakes.

conditions were poor for the measurement of underwater light, with considerable wind ripple on the lake surface, and broken cloud cover causing incident radiation to fluctuate markedly and continuously. This weather caused some error in recording irradiance, evidenced by the kinks in some of the profiles, and prevented the measurement of the spectral distribution of P.A.R. in the lakes.

Euphotic depths could only be directly measured in six lakes (see Table 3.8). In the others, euphotic depths were greater than the actual depth at the sampling point. The shallowest of those measured were from Red Lagoon (0.73 metres) and Lake Cooloomera (1.21 metres), the two most dystrophic lakes, and from Brown Lake (1.42 metres), which had the highest turbidity and chlorophyll a content.

$Kd_{(av)}$ and $Ku_{(av)}$ were mostly similar in those lakes where upwelling irradiance was measured, although $Ku_{(av)}$ was usually less. The greatest differences between the two were in Lake Wabby and Blue Lake, where $Ku_{(av)}$ was only 77.1% and 76.2% of $Kd_{(av)}$, respectively. Some enhancement of upwelling irradiance by reflection of light from off the bottom may have occurred in these lakes, as euphotic depth exceeded actual depth. $Ku_{(av)}$ was more than $Kd_{(av)}$ in humic Lake Boomanjin, due possibly to inaccuracies in measuring the low levels of upwelling irradiance, which was so weak in highly dystrophic Lake Cooloomera and Red Lagoon that measurement was impossible, even close to the surface.

Reflectance profiles varied considerably between lakes (Figure 3.14). These too could be influenced by reflection from the bottom or low irradiance measurements. Reflectance was greatest in Lake Wabby, despite its low turbidity, and least in humic lakes such as Coomboo, Boomanjin, and Hidden Lakes. Three types of profiles were apparent. Firstly, reflectance increased continuously with depth, as in Lake Wabby, although sometimes in a rather haphazard manner (e.g. Brown Lake). The second type, examplified by White Lake (and to a much less extent by Basin Lake), was where reflectance peaked after an initial increase, and then declined. The third type was where very low reflectance values stayed virtually constant with depth, as in Lake Boomanjin and Hidden Lake. Some fluctuations within the profiles may be due to rapid changes in the intensity of the incident light field during measurement.

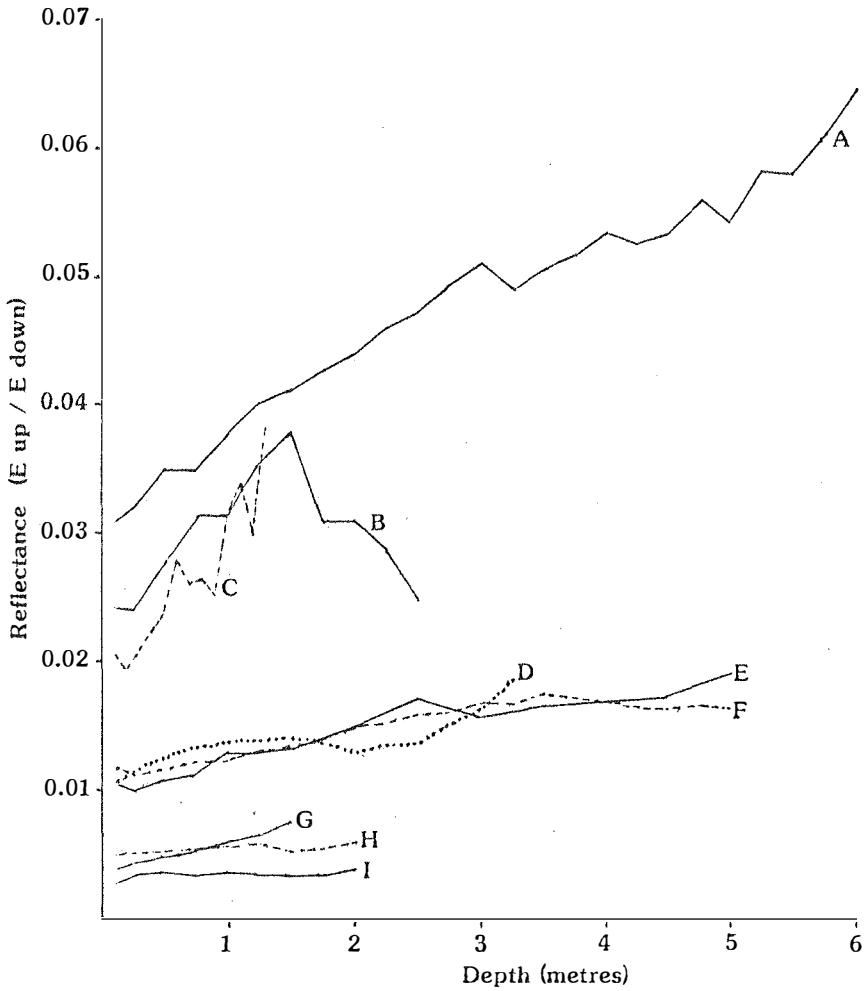


Figure 3.14 Reflectance/depth profiles for selected south-east Queensland coastal lakes, April/May, 1987. A. Lake Wabby; B. White Lake; C. Brown Lake (dashed line); D. Ocean Lake (dotted line); E. Blue Lake, North Stradbroke Island; F. Basin Lake (dashed line); G. Lake Coomboo; H. Lake Boomanjin (dashed line); I. Hidden Lake.

Measurements of the scattering coefficient, b, could only be made on nine of the lakes. Values ranged from 6.32 m^{-1} in Brown Lake to 0.58 m^{-1} in Hidden Lake, and most were higher than anticipated for non-turbid waters.

3.3.20 Secchi Disc Transparency

Water transparency, as measured by the Secchi disc, is also indicated in Table 3.8. In half the lakes, this exceeded the depth of the bottom at the sampling point. The minimum value was 0.65 metres in polyhumic Red Lagoon, and the deepest was in Lake McKenzie, where the Secchi disc was still visable lying on the bottom, 7.75 metres deep.

3.3.21 Regression and Correlations

The results of linear and multiple regression analyses of turbidity, gilvin, and chlorophylla against $Kd_{(av)}$ are shown in Table 3.9. Gilvin was the main factor affecting light attenuation and determining $Kd_{(av)}$ in the waters of these lakes, while turbidity and particularily chlorophyll a were of much less importance. Both turbidity and gilvin together as the independent variables (Equation 4) improve the prediction of Kd(av) slightly, but further inclusion of chlorophyll a in the multiple regression does not. Log normal transformation of the data made no improvement of fit for these regressions, and so are not shown.

Use of Pearson correlation analysis shows several of the parameters to be strongly correlated. Water colour was negatively correlated to pH ($r = -0.5233$, $P = 0.006$), with pH decreasing as the humic content of the water rises, as shown previously by Bayly (1964). Due to this, and the strong influence of colour on the vertical attenuation coefficient for downwelling irradiance, a significant positive correlation was also recorded between pH and $Kd_{(av)}$. pH also correlated significantly with K_{18} ($r = 0.6679$, $P \ll 0.001$), supporting the suggestion (Section 3.3.16) that buffering capacity was increased in those lakes with higher ionic concentrations.

There were no significant correlations between $Kd_{(av)}$ and either turbidity or chlorophyll a (see Table 3.9). However, turbidity was significantly positively correlated with chlorophyll a ($r = 0.6652$, $P \ll 0.001$), indicating that much of the particulate matter in these lakes could be phytoplankton, rather than tripton. Additionally, total

Table 3.9: Regression analysis showing the relationships between various optical parameters in the freshwater coastal dune lakes of south-east Queensland

Equation No.		r^2	n	P
1	$Kd_{(av)} = 0.681 + 0.171\ g_{440}$	.805	26	$\ll 0.001$
2	$Kd_{(av)} = 0.491 + 0.837\ Tn$	.149	26	0.052
3	$Kd_{(av)} = 1.04 + 0.035\ Chla$	.015	25	0.562
4	$Kd_{(av)} = 0.062 + 0.73\ Tn + 0.167\ g_{440}$	.918	26	$\ll 0.001$
5	$Kd_{(av)} = 0.372 + 0.175\ g_{440} + 0.061\ Chla$	.851	25	< 0.001
6	$Kd_{(av)} = 0.493 + 1.21\ Tn = -0.071\ Chla$	.189	25	$0.20 < P < 0.05$
7	$Kd_{(av)} = 0.056 + 0.776Tn + 0.166g_{440} - 0.008Chla$	.919	25	< 0.001
8	$\underline{b} = 2.26\ Tn - 0.788$	.935	9	< 0.001

phosphorus was significantly positively correlated with both turbidity ($r = 0.5512$, $P = 0.004$) and chlorophyll a ($r = 0.5996$, $P = 0.001$), but total nitrogen was correlated significantly only with g_{440} ($r = 0.5259$, $P = 0.006$).

3.3.22 Phytoplankton

Phytoplankters which were abundant or common in at least one location are given in Table 3.10. Because their taxonomy was, for the most part, taken only to genus level, only thirty groups are given, but many more would have resulted had it been possible to identify everything to species level. Other species may have been missed by the 20 µm mesh plankton net employed for sampling.

The two main components were the desmids and the dinoflagellates. *Peridinium* was absent only from Red Lagoon, and no desmids were present in Blue Lake, North Stradbroke Island. At least four different species of *Peridinium* were identifiable, with *P. granulosum* Playfair being the most common, and present in almost every lake. There were also a number of species of *Staurastrum* and *Staurodesmus*. *Dynobryon cylindricum* Imhof was present in many of the lakes, but *D. sertularia* Ehrenberg was also present in large numbers in Ocean Lake. A small, green coccoic colonial alga of the *Sphaerocystis* type was also common or abundant in a number of lakes, but the remaining taxa were usually present in small numbers only.

At the taxonomic level given, there appears to be little difference between the species diversity of those lakes with the highest nutrient and chlorophyll a concentrations, and that of the more oligotrophic lakes, or between the least attenuating and the more highly attenuating waters.

3.4 DISCUSSION

3.4.1 Factors Influencing the Underwater Light Climates of the Three Study Areas

Although water itself may be a significant attenuator in the clearest lakes, dissolved humic substances constitute the most important component of the aquatic medium influencing the underwater light climates of all three study areas. Many Tasmanian lakes, especially those from the western half of the island, are considerably dystrophic, more so than those from elsewhere in Australia (Table 1 in Kirk, 1986),

Table 3.10: Phytoplankton from the freshwater coastal lakes of south-east Queensland. A = abundant, C = common, x = present in small numbers.

Lake No.	LAKE \ ALGA	*Gonium* sp.	*Asterococcus superbus* (Cienk.) Scherf.	cf. *Sphaerocystis*	*Ankistrodesmus falcatus* (Corda) Ralfs.	*Kirchneriella lunaris* (Kirchner) Möbius.	*Nephrocytium lunatum* W. West.	*Oocystis* sp.	*Botryococcus braunii* Kützing	*Mougeotia* sp.	*Spirogyra* sp.	*Cylindrocystis* sp.	*Arthrodesmus octocornis* Ehrenberg	*Closterium* sp.	*Cosmarium* spp.	*Desmidium* spp.	*Pleurotaenium* sp.	*Staurastrum* spp.	*Staurodesmus* spp.	*Triploceros gracile* Bailey	*Xanthidium* spp.	*Euglena* sp.	*Trachelomonas* spp.	*Peridinium* spp.	*Dynobryon* spp.	*Rhizosolenia eriensis* H.L. Smith	*Surirella spiralis* Kuetzing	Ovoid diatom (?)	*Merismopedia* sp.	*Microcystis aeruginosa* Kützing emend. Elenkin	*Anabaena* sp.
1	Lake Minker	C			X		X	X							A			A	C				C	A							A
2	Lake Woonjeel			X					C						X			X	X			X		A	C						
3	Ocean Lake			X	X	C	C	C	C	C	A							X				C	X	C	A					X	X
4	White Lake							X	X						X			A	A					C	C	C	X				
5	Lake Bowarrady			X	C							C		X	X		X	C	A	X	X			A	C				X	X	X
6	Lake Allom	X	X	A				X	C						X			X						A			X				X
7	Deepwater Lake				X										X		X	C	X	X				A	C						X
8	Lake Coomboo								X	X		C	C		X		X	A		X	X			A	A		X		X	C	X
9	Hidden Lake			A					A			C						C						A			X				
10	Boomerang Lake (north)																	X	C		X			A	X	X	X			X	
11	Boomerang Lake (south)		C	C				X	C			X			X		X	C		X	X		X	X	X					X	
12	Lake Garawongera			C				X	X		X	X		X	C	C	C	A	A	C	X	X		A	C	X					X
13	Lake Wabby			A		X	C	X	X	X					A			A		X			X	C	A	X					X
14	Lake McKenzie		X	A	X			A		X							X	X						A	X		X		X		
15	Basin Lake			A					X			X			C	A		A	A	X				C			X				
16	Jennings Lake			X					A			X			C	X	X	C	X	C				A	X		C				X
17	Lake Birrabeen		X	A	X					C			X		X		C	X	X	X			X	A	X		X		X		
18	Lake Benaroon			C								X			C		X	C	C	C	X			A			C		X	X	X
19	Lake Boomanjin			X					X			X	X		X		X	C	A		C			C	X	C	X				
20	Red Lagoon			X					X			A						C							X		X				X
21	Lake Freshwater									X							X	X	X				X	A	X						
22	Lake Cooloomera								X								X	X	X			C		A	X						
23	Blue Lagoon							X				X						X	A	X	X			A	X		X				
24	Honeyeater Lake									A			X				X	C	A				C	A	X	X					X
25	Brown Lake			A				X	X									C				X		X			X				
26	Blue Lake			X																				A	X	A	X	C			X

Chlorophyceae

Chrysophyceae
Euglenophyceae
Bacillariophyceae
Dinophyceae

Cyanophyceae

or from New Zealand (Vant and Davies-Colley, 1984). Gilvin concentrations in the lentic freshwaters of north-east New South Wales and the coastal dune lakes of south-east Queensland were at only slight to moderate levels, but this was still the most significant attenuator of P.A.R. in the majority of these waters also. Gilvin too is responsible for the greater part of light attenuation in inland waters from south-east New South Wales and the Australian Capital Territory (Kirk, 1976b, 1977b, 1981c), and north Queensland (Finlayson, et al, 1984).

The Queensland coastal dune lake waters, and most of the Tasmanian lakes and reservoirs studied were of low turbidity, and even the more turbid of them generally had lower concentrations of suspended particulate matter than many inland waters from elsewhere in Australia (Table 1 in Kirk, 1986), and some from New Zealand (Vant and Davies-Colley, 1984). The study sites of north-east New South Wales were generally more turbid, but even these had for the most part low turbidity. Therefore, while being of local importance such as in Kentucky Creek Dam and Yarrie Lake in New South Wales, and in Tooms Lake, Tasmania, overall tripton plays only a small role in the extinction of underwater light in these areas, its effects being surpassed by gilvin. Elsewhere however, such as in Gippsland, South Australia, and the Northern Territory, turbidity can be the major attenuator , at least seasonally (Hickman et al, 1984; Ganf, 1980; Geddes, 1984; Walker, 1984; Kirk and Tyler, 1986).

Phytoplankton are considered significant attenuators of P.A.R. only at chlorophyll a concentrations exceeding 10 $\mu g\ L^{-1}$ (Talling, 1960; Kirk, 1977b). Since the majority of Tasmanian lakes are oligotrophic (Tyler, 1974), phytoplankton is therefore unlikely to contribute much to light attenuation within them, and the same is true for most coastal dune lakes of south-east Queensland. However, phytoplankton would probably significantly attenuate light in some of the eutrophic north-east New South Wales reservoirs, especially Quipolly, St. Helena, Dumaresq, and Puddledock Dams, in a manner comparable to that in Solomon Dam, Palm Island, Queensland (Hawkins and Griffiths, 1986).

The lakes and reservoirs covered in this study represent a range of clarities. However, values of the mean vertical attenuation coefficients, especially those for Tasmania and south-east Queensland, fell mostly towards the lower half of the range reported for other

Australian waters (Kirk, 1977b, and Table 1, 1986). The clearwater coastal dune lakes of Queensland are usually considered to be of the water-table window type (Timms, 1982), and have $Kd_{(av)}$ values similar to many of the non-dystrophic montane lakes of Tasmania, and New Zealand (Vant and Davies-Colley, 1984; Howard-Williams and Vincent, 1984). In comparison, the $Kd_{(av)}$ values of the lentic waters of north-east New South Wales correspond to the upper halves of the ranges reported for dystrophic Tasmanian lakes, and those from New Zealand (Vant and Davies-Colley, 1984; Howard-Williams and Vincent, 1984), being instead similar to some Northern Territory billabongs (Kirk and Tyler, 1986) and Mt Isa, Queensland, reservoirs (Finlayson et al, 1984). The results of this study of north-east New South Wales waters are in close agreement with the measurements of attenuation made previously on some reservoirs from the region by Scribner (quoted in Kirk, 1986), and the attenuation of broadband P.A.R. in Blue and Brown Lakes, North Stradbroke Island, Queensland, fell within the ranges estimated for these two lakes from the results of Bensink and Burton (1975), and Lee-Manwar (unpublished) (0.30 to 0.59 m^{-1} for Blue Lake; 1.28 to 7.09 m^{-1} for Brown Lake).

The strong influence of gilvin in determining the underwater light fields of inland waters can be seen even in the clearest lakes, such as Lakes Perry, Laura, St. Clair, and Great Lake, in Tasmania, and Lake Hiawatha in New South Wales. Although gilvin levels in these lakes are at the threshold of detection by the laboratory methods employed in this study, there is still sufficient present to cause the perceptible attenuation of blue light in these waters at depth, resulting in spectral distributions centred on the green to yellow wavelengths. The non-turbid, non-humic oligotrophic Queensland lakes would have similar light climates, as shown by measurements from Blue Lake, North Stradbroke Island, obtained by use of coloured filters on a broadband sensor (Bensink and Burton, 1975; Lee-Manwar, unpublished). The spectral distributions and irradiance profiles from the clear-water lakes of this study are more akin to those of coastal waters off south-east Australia (Kirk, 1977a, 1979). Clearer inland waters however, with considerably lower values of $Kd_{(av)}$ (as low as 0.10 to 0.12 m^{-1} for Lake Coleridge) exist in New Zealand (Vant and Davies-Colley, 1984; Howard-Williams and Vincient, 1984), and there was no equivalent of extremely pure Crater Lake, Oregon, where blue light dominates the P.A.R. spectrum at depth (Smith and Tyler, 1967; Smith et al, 1973).

The rate of increase of attenuation, and concomitant decrease in euphotic depth, is greatest at initially low gilvin levels, but decreases as gilvin concentration increases (Canfield and Hodgson, 1983; Eloranta, 1978; Kirk, 1976b). The spectral distribution of underwater P.A.R. also changes as humic content increases, with the wavelength of maximum tranmission moving from the green region of the spectrum towards the red, along with an associated narrowing of the transmission window. In highly coloured dystrophic lakes the influence of gilvin is extreme, with rapid and almost complete extinction of blue light within the first metre of water. Gilvin at high concentrations also absorbs green, yellow, and even some orange light (see for examples the absorption spectra of gilvin in Figure 3.9, and in Figure 4.4), leaving narrow underwater spectral distributions centred on red light, at 650 to 700 nm. This is shown clearly by the P.A.R. spectra for Lake Ainsworth, New South Wales, and Lakes Pedder (nova) and Chisholm, Tasmania, as well as for other highly dystrophic lakes from the Gordon River area of south-west Tasmania (King and Tyler, 1981 b, 1982a, 1983; Croome and Tyler, 1984a, 1985a; Bowling and Tyler, 1986). Dystrophic Queensland lakes such as Lakes Coomboo and Cooloomera, and Red Lagoon, would have similar underwater light climates. The effect of gilvin on the spectral bandpass of downwelling P.A.R. is summarised in Figure 3.15a, where data from the measured euphotic depth are shown for a series of selected Tasmanian lakes of increasing gilvin concentration.

While tripton undoubtedly contributes something to light attenuation in all but the clearest waters, it was a major contributor in only a few locations, even in north-east New South Wales. The effect of increasing turbidity, summarised for a series of increasingly turbid Tasmanian lakes in Figure 3.15b is identical to that of increasing gilvin concentration, causing increased attenuation, decreased euphotic depths, and a shift in the spectral distribution of P.A.R. with depth, from being centred on green light at low tripton concentrations, to being centred on red light when they are high. The almost identical underwater spectra shown by turbid lakes such as Lake Crescent and Tooms Lake, Tasmania, and Yarrie Lake, New South Wales, to those of non-turbid, humic lakes, suggests that the attenuation pattern produced by suspended matter, especially tripton, results less from its scattering effect than from absorption by humic materials either bound to the outside of the mineral particles, or existing as free particles

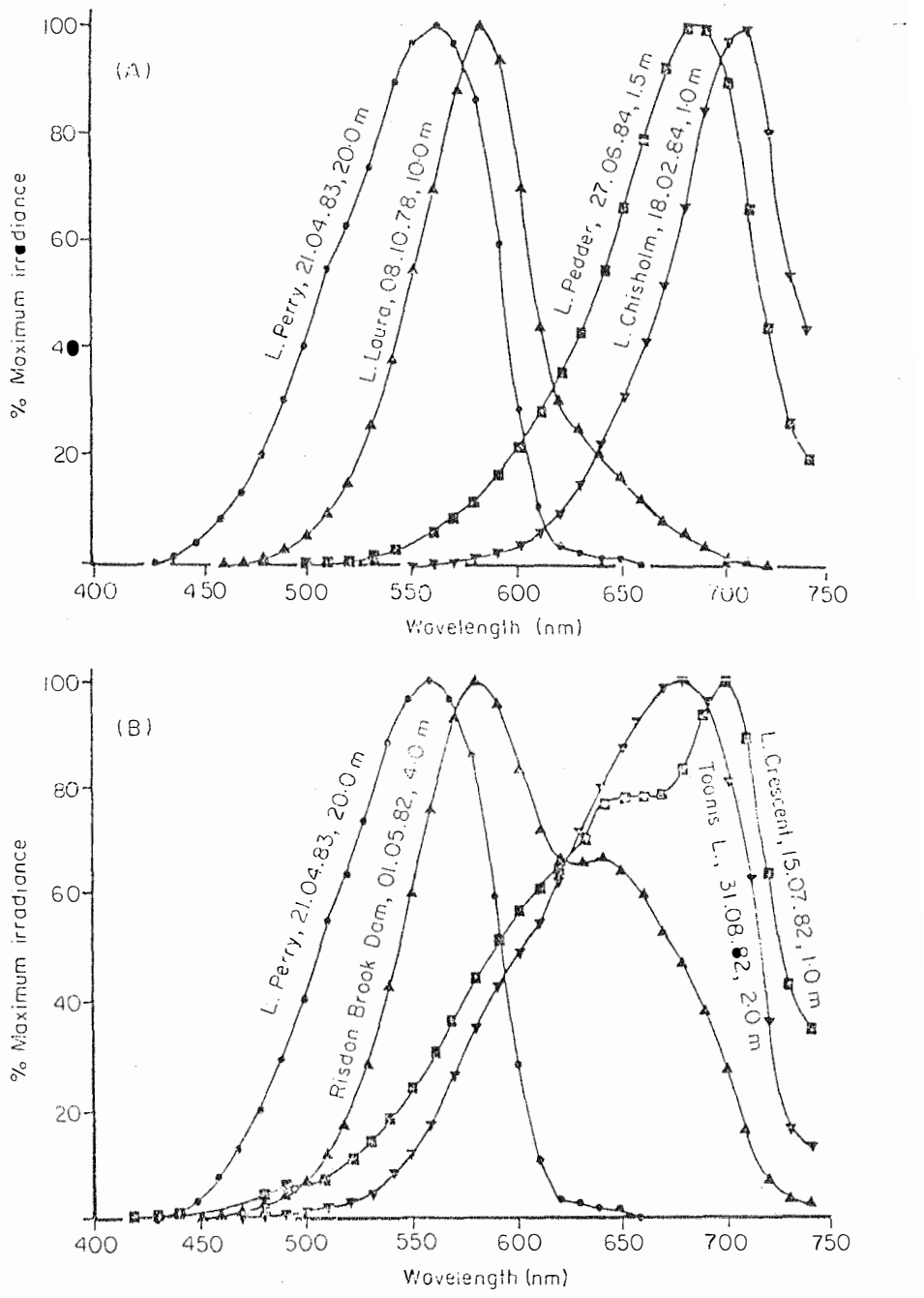


Figure 3.15 Effect of gilvin and turbidity on the spectral bandpass of downwelling P.A.R. at the measured euphotic depth for selected Tasmanian lakes. (A) A series of increasing gilvin concentrations. (B) A series of increasing turbidity. For ease of comparison, in each case values are plotted as percentage of the irradiance at the most penetrating wavelength. Values for gilvin and turbidity can be obtained from Table 3.2.

(Kirk, 1980a; 1983). The absorption spectra of particulate matter have much the same shape as those of dissolved humic substances (Kirk, 1980a; Davies-Colley, 1983).

The spectral distribution and attenuance of irradiance in turbid Tasmanian lakes is comparable to that measured on occasions in Lakes Burley Griffin and Ginninderra, from the Australian Capital Territory (Kirk, 1979), some Northern Territory billabongs (Walker, 1984, Walker and Tyler, 1984; Kirk and Tyler, 1986), and in Rust der Winter Dam, South Africa (Walmsley et al, 1980), where turbidity was also the major attenuating factor. The extreme turbidity of Yarrie Lake, New South Wales caused a similar underwater spectrum, composed almost entirely of red light, but, unlike the others, its wavelength of maximum tranmission exceeded 700 nm, resulting from its much greater tripton loading. Its downwelling vertical attenuation coefficient also exceeded any others measured in highly turbid Australasian waters, including Lake George, New South Wales (Kirk, 1977b); Lake Alexandrina, South Australia (Geddes, 1984); Georgetown Billabong, Northern Territory (Kirk and Tyler, 1986), and Lake Wairarapa, New Zealand (Howard-Williams and Vincent, 1984).

Phytoplankton may influence the underwater light climates of eutrophic lakes, not only removing blue light (along with gilvin and tripton), but also causing noticable absorption troughs in the vicinity of 670-680 nm, as shown by the spectral distributions of P.A.R. from Lake Sorell in Tasmania and Dumaresq Dam in New South Wales. Similar absorption troughs due to phytoplankton chlorophyll a have been reported from Lake Burley Griffin, Australian Capital Territory (Kirk, 1979); Island Billabong, Northern Territory (Kirk and Tyler, 1986); Lake Kinneret, Israel (Dubinsky and Berman, 1979), and Lough Neagh, Northern Ireland (Jewson, 1977). Phytoplankton were present in sufficient quantity in Brown Lake, Queensland, to also modify its underwater light climate, so it too would probably have a spectral distribution of P.A.R. similar to Dumaresq Dam. Stratification of phytoplankton in Storm King Dam noticably increased P.A.R. attenuation below 2.0 metres in this reservoir, and had modified its spectral distribution at 3.0 metres to include a marked chlorophyll a absorption trough at 680 nm. Stratified phytoplankton and photosynthetic bacteria cause similar heterogeneous light climates in meromictic Tasmanian lakes (Bowling and Tyler, 1986).

The spectral distributions of a number of lakes with only slight levels of gilvin and turbidity (e.g. around 1.000 m^{-1} and 2.0 N.T.U.) showed a broad transmission plateau of wavelengths between about 590 to 660 nm, rather than a sharp peak of maximum transmission. A slight absorption trough, centred at 620 nm, bisected this plateau in some, including Lake Leake and Risdon Brook Dam from Tasmania, and Lake Minnie Water, Lake Keepit, and Chaffey Dam from New South Wales, and was present as a shoulder on spectral scans from others. Similar troughs and shoulders were also present in spectral scans from Lakes Burley Griffin and Ginninderra from the Australian Capital Territory, and Burrinjuck Dam in south-east New South Wales (Kirk, 1979). No explanation can be offered to account for the slightly higher absorption of light at these wavelengths. Phycocyanin contained in blue-green algae created an absorption trough close to 620 nm in spectral distributions from Lough Neagh, Northern Ireland (Jewson, 1984). However the low chlorophyll a contents of the study sites, plus the absence of chlorophyll a attenuation troughs at 680 nm in the spectral scans, indicates that phytoplankton were present in insufficient quantities to appreciably attenuate light. Additionally, cyanobacteria constituted only a small part of the phytoplankton flora of these lakes, or were entirely absent.

Although statistically meaningful relationships were obtained between Secchi disc depth and euphotic depth for Tasmanian (Table 3.3) and New South Wales (Table 3.6) waters (insufficient data were available for Queensland lakes), care should be exercised in the use of such empirical relationships. The results from these two studies suggest that different attenuating components of the aquatic medium had differing effects on Secchi disc depth. This is especially true for non-turbid dystrophic waters, where the Secchi disc is often visible below the measured euphotic depth.

Variations in the prediction of euphotic depth, and vertical attenuation coefficients ($Kd_{(av)}$) from Secchi disc measurements result from these being dependant not only on $Kd_{(av)}$, but also on the beam attenuation coefficient ($\underline{c}$), and these may vary independantly of each other to some extent (Tyler, 1968).

This study was restricted to one sampling per waterbody in north-east New South Wales. The limited data of Scribner (quoted by Kirk, 1986) indicate that considerable seasonal and temporal fluctuations occur in the attenuating components and underwater light climates of

these waters, as they do in many other Australian inland fresh waters (Kirk, 1977b, 1979, 1986; Hickman et al, 1984; Ganf, 1980; Geddes, 1984; Walker, 1984; Walker and Tyler, 1984; Finlayson and Gillies, 1982; Finlayson et al, 1984; Hawkins and Griffiths, 1986). Therefore, any extent of seasonal change, and the variability this may have caused between sites sampled at different times during the study, cannot be assessed. However, Scribner's data (quoted by Kirk, 1986) indicate temporal changes of much smaller magnitude for the waters of this area compared to those which occur in Northern Territory billabongs (Walker, 1984; Walker and Tyler, 1984), or reported for Nebraska reservoirs, one of which varies over a thirty-four fold range (Roemer and Hoagland, 1979).

Measurements of various optical parameters from Blue and Brown Lakes, North Stradbroke Island (Bensink and Burton, 1975; Lee-Manwar, unpublished), indicate similar temporal variation is likely in the freshwater coastal dune lakes of south-east Queensland, and is probably why Secchi disc depths for many lakes of this study exceeded those reported previously (Bayly et al, 1975; Bensink and Burton, 1975; Miller et al, 1976, 1984; Arthington et al, 1986). With greater Secchi disc depths, the lakes were probably considerably less humic than during earlier studies, but there are no published values of colour as either g_{440} or as Hazen (mg Pt L^{-1}) with which to compare those of this survey. Previously only optical densities at 385 nm have been reported (Bayly, 1964; Bensink and Burton, 1975), and although giving an indication of which lakes were more humic than others, make comparisons impossible.

Although data on seasonal variation in Tasmanian lakes is limited, dystrophic lakes have been shown to be humic all year (King and Tyler, 1981a,b, 1982a, 1983; Steane and Tyler, 1982); the few turbid lakes always turbid (Cheng and Tyler, 1973a; Croome and Tyler, 1972), and the clearwater lakes are always so (Tyler, 1967). (See also Table 3.2 for previously published ranges of colour and turbidity in some Tasmanian lakes). Such seasonal variation as does occur in Tasmanian inland waters, such as changes in gilvin concentration in the dune lakes of the west and south-west coasts (see Chapter Four), are likely to produce variations of degree, rather than kind, in the underwater light climate. The greatest variation in Tasmanian waters occurred in Risdon Brook Dam, an urban water storage, where water is added sporadically from a number of sources by the management authority.

3.4.2 Upwelling Irradiance, Reflectance, and Scattering

Upwelling irradiances were attenuated in much the same way with depth as downwelling irradiance in the waters from all three study areas, as found also by Kirk (1977b) for south-east New South Wales waters. However, the attenuation coefficient for upwelling irradiance, $Ku_{(av)}$, was in most cases slightly less than the corresponding coefficient for downwelling P.A.R. This is because measurements of upwelling irradiance are made on light that has been spectrally modified by removal of the more readily absorbed wavelengths (see Figure 3.8), with the upwelling light stream therefore containing a proportionally greater amount of less readily attenuated wavelengths than does downwelling P.A.R. at the same depth. The greatest variations between $Ku_{(av)}$ and $Kd_{(av)}$ occurred in some of the shallower clear-water lakes, such as Ladies Tarn in Tasmania and Lake Wabby in Queensland, possibly through the enrichment of upwelling irradiance by reflection from off the bottom, and in highly humic lakes like Lake Pedder (nova) (Tasmania) and Lake Boomanjin (Queensland), where low levels of upwelling irradiance make accurate measurement difficult.

The majority of surface reflectance values, R(o), calculated for the inland waters of north-east New South Wales fell towards the lower end of the range listed for other Australian waters by Kirk (1986), and from New Zealand (Howard-Williams and Vincent, 1984). Surface reflectance values measured in Tasmanian and south-east Queensland lakes were generally even lower than those from north-east New South Wales. Reflectance near the surface was highest in the turbid lakes, but the non-turbid, humic lakes of all three study areas had extremely low R(o) values.

Changes in reflection with depth followed one of three forms. In many north-east New South Wales waters, and some Queensland coastal lakes, it increased continuously with depth. This type of profile has also been recorded in transparent New Zealand lakes (Howard-Williams and Vincent, 1984), where reflection from the bottom increased the ratio of upwelling irradiance to that of downwelling with increasing depth. However, this is unlikely for many of the deeper, turbid lakes, where these profiles occurred in this study. In others, such as White Lake, Queensland; St. Helena Dam, New South Wales, and Lake Sorell and Tooms Lake, Tasmania, reflectance decreased following an initial increase with depth. Jewson (1984) also found similar curved reflection profiles in two Austrian lakes with suspended calcite

particles. Only in a few lakes in New South Wales and Queensland did reflection values level off to approach a constant value with depth, producing profiles similar to those for reservoirs in south-east New South Wales and the Australian Capital Territory (Kirk, 1977b), some New Zealand lakes (Howard-Williams and Vincent, 1984), and Lough Neagh, Northern Ireland (Jewson, 1984). No explanations can be made for the variations from this theoretical (Kirk 1977b, 1983) asymptotic depth-reflectance curve in most of the waters of this study.

The scattering coefficients, $\underline{b}$, calculated for the three study areas were consistant with the ranges reported elsewhere for Australia and New Zealand (Kirk, 1981a, 1986; Kirk and Tyler, 1986; Phillips and Kirk, 1984; Vant and Davies-Colley, 1984), although those from Tasmania and Queensland were for the most part lower than those from north-east New South Wales, being instead similar to those calculated for the coastal waters of Jervis Bay (Phillips and Kirk, 1984) and for non-turbid New Zealand lakes (Vant and Davies-Colley, 1984). That for Yarrie Lake is higher than any previously recorded. An approximate 1:1 ratio between $\underline{b}$ and nephalometric turbidity has been found in a number of diverse locations (Di Toro, 1978; Kirk, 1981a; Kirk and Tyler, 1986; Davies-Colley, 1983), and many of the waters of north-east New South Wales also fit this pattern. The range in the $Tn/\underline{b}$ ratio for this area was 0.50 to 1.50, with a mean of 0.89±0.23. This does not hold for the Tasmanian lakes however, where the range of $Tn/\underline{b}$ was 0.29 to 1.22, with a mean of 0.68±0.23. The ratios between turbidity and the scattering coefficients for the nine coastal freshwater lakes of south-east Queensland where $\underline{b}$ was measured were highly variable, ranging between 0.46 to 1.59 ($\bar{x}$ = 0.80±0.37). The waters with the lowest $Tn/\underline{b}$ ratios from all three study areas were those with the lowest turbidity values; and these were considerably lower than the turbidity values given for the waters studied by Di Toro (1978), Kirk (1981a), Kirk and Tyler (1986), and Davies-Colley (1983). Thus, it is suggested that the accuracy of measurement of $\underline{b}$ by Kirk's (1981a) method may become more precise at higher levels of turbidity, and therefore scattering. Regression analyses from each of the three study areas (Tables 3.3, 3.6, 3.9) indicate that turbidity is the major factor contributing to the scattering coefficient. The turbidity to scattering coefficient ratio for twenty-five New Zealand lakes, many of them turbid, calculated from data given in Vant and Davies-Colley (1984), also reveal discrepencies from 1.0 (range 0.33 to 1.88, $\bar{x}$ = 0.85±0.37).

3.4.3 Humics, Turbidity, and Chlorophylla

Kirk (1980a) proposed that the origin of both the suspended particulate fraction and dissolved humic substances in turbid, yellow Australian inland waters was from the soils of the catchment. If so, both gilvin and tripton would be washed into a lake or reservoir together, and concentrations of both should rise and fall concurrently. Such interactions would explain the significant positive correlation between gilvin and turbidity found in the north-east New South Wales waters of this study, although this was somewhat weaker than the one ($r = 0.80$) reported by Kirk (1979) for south-eastern New South Wales and Australian Capital Territory waters. Here levels of both increase after heavy rainfall and consequent soil erosion, although not exactly in parallel. However, the high levels of dissolved organic colour in the non-turbid, dystrophic lakes of Tasmania and south-east Queensland must originate from other sources, being derived especially from decaying vegetation in the catchment, and then dissolved and washed into the lakes by surface runoff after rainfall. Such a mechanism has been proposed for the presence of dissolved humic substances in New York lakes (Cronan and Aiken, 1985).

Coloured organic materials may also be derived from photosynthetic microorganisms within the water column, through their excretion and decomposition. Although no significant correlation was found between g_{440} and chlorophyll a, it is possible that some of the dissolved humics in the eutrophic freshwaters of north-east New South Wales with high chlorophyll a concentrations, such as St. Helena Dam, are produced autochthonously by their phytoplankton in this manner. A similar source of colour has been suggested for some highly eutrophic and productive New Zealand lakes (Vant and Davies-Colley, 1987), and in highly stratified meromictic Tasmanian lakes, where both g_{440} and photosynthetic microorganisms increase markedly together with depth (Baker et al, 1985a; Bowling and Tyler, 1986). However, this would be an unlikely source of gilvin in Tasmanian and Queensland lakes of this study, as most are considered oligotrophic and unproductive (see Sections 2.1 and 2.3.2).

3.4.4 Optical Classifications of the Lakes of the Study Areas

Cluster analyses were undertaken to classify the lakes of the study areas in terms of their optical characteristics and light climates, based on the measured gilvin and turbidity values for the lakes.

Tasmania's lakes fall readily into a series of categories (Figure 3.16). Group I contains lakes of low colour ($\bar{x}$ = 0.626 m^{-1}, σ = 0.41) and turbidity ($\bar{x}$ = 0.69 N.T.U., σ = 0.44). From regression Equation 5 for Tasmanian waters (Table 3.3), these waters could be expected to have average downwelling attenuation coefficients of around 0.75 m^{-1}, and underwater spectral distributions of P.A.R. similar to those of Lake Laura (Figure 3.7a) or Lake Leake (Figure 3.7b). Group II contains lakes with slightly higher colour ($\bar{x}$ = 2.316 m^{-1}, σ = 1.13) or turbidity ($\bar{x}$ = 1.49 N.T.U., σ = 0.36); would have $Kd_{(av)}$ values of approximately 1.13 m^{-1}, and spectral distributions close to those of Risdon Brook Dam (Figure 3.7b), or Lake Barrington on 13-3-78 (Figure 3.7d). There was little distance between these two groups in the cluster analysis.

Lakes with moderate colour ($\bar{x}$ = 5.70 m^{-1}, σ = 1.45) but low turbidity ($\bar{x}$ = 0.82 N.T.U., σ = 0.52) account for Group III. These would have average downwelling attenuation coefficients of about 1.66 m^{-1}, and have spectral distributions of P.A.R. at depth centred around 670-680 nm, as in Lakes Curly and Wurrawina (Figure 3.7d). Conversely, Group IV contains those lakes of moderate turbidity ($\bar{x}$ = 4.35 N.T.U., σ = 0.07) but low colour ($\bar{x}$ = 0.72 m^{-1}, σ = 0.69). The expected $Kd_{(av)}$ for this type of lake would approximate 1.14 m^{-1}, and it would have spectral distributions similar to those typified by Lake Sorell (Figure 3.7c), where blue light is rapidly attenuated, but a broad transmission plateau between 575 and 700 nm remains.

The highly coloured lakes ($\bar{x}$ = 10.18 m^{-1}, σ = 3.62) with low turbidity ($\bar{x}$ = 0.46 N.T.U., σ = 0.21) make up Group V, while the few lakes of lower colour ($\bar{x}$ = 4.03 m^{-1}, σ = 1.68) but considerable turbidity ($\bar{x}$ = 11.38 N.T.U., σ = 4.25) fall into Group VI. Their anticipated $Kd_{(av)}$ values would be around 2.42 m^{-1} and 2.45 m^{-1}, respectively. The typical spectral distribution of underwater P.A.R. for Group V would be like those of Lake Pedder on 27-6-84, and Lake Strahan (Figure 3.7e), while that for Group VI would be very similar, resembling those from Lake Crescent and Tooms Lake (Figure 3.7c). Lake Chisholm, a polyhumic lake ($\bar{x}$ = 25.59 m^{-1}, σ = 1.38) of low turbidity ($\bar{x}$ = 0.54 N.T.U., σ = 0.15) on all occasions stood apart as the sole representative of Group VII.

The classification of lake types so obtained agrees favourably with one based on edaphic and chemical data by Buckney and Tyler (1973a). Thus, the majority of alpine and lowland lakes of Tasmania's

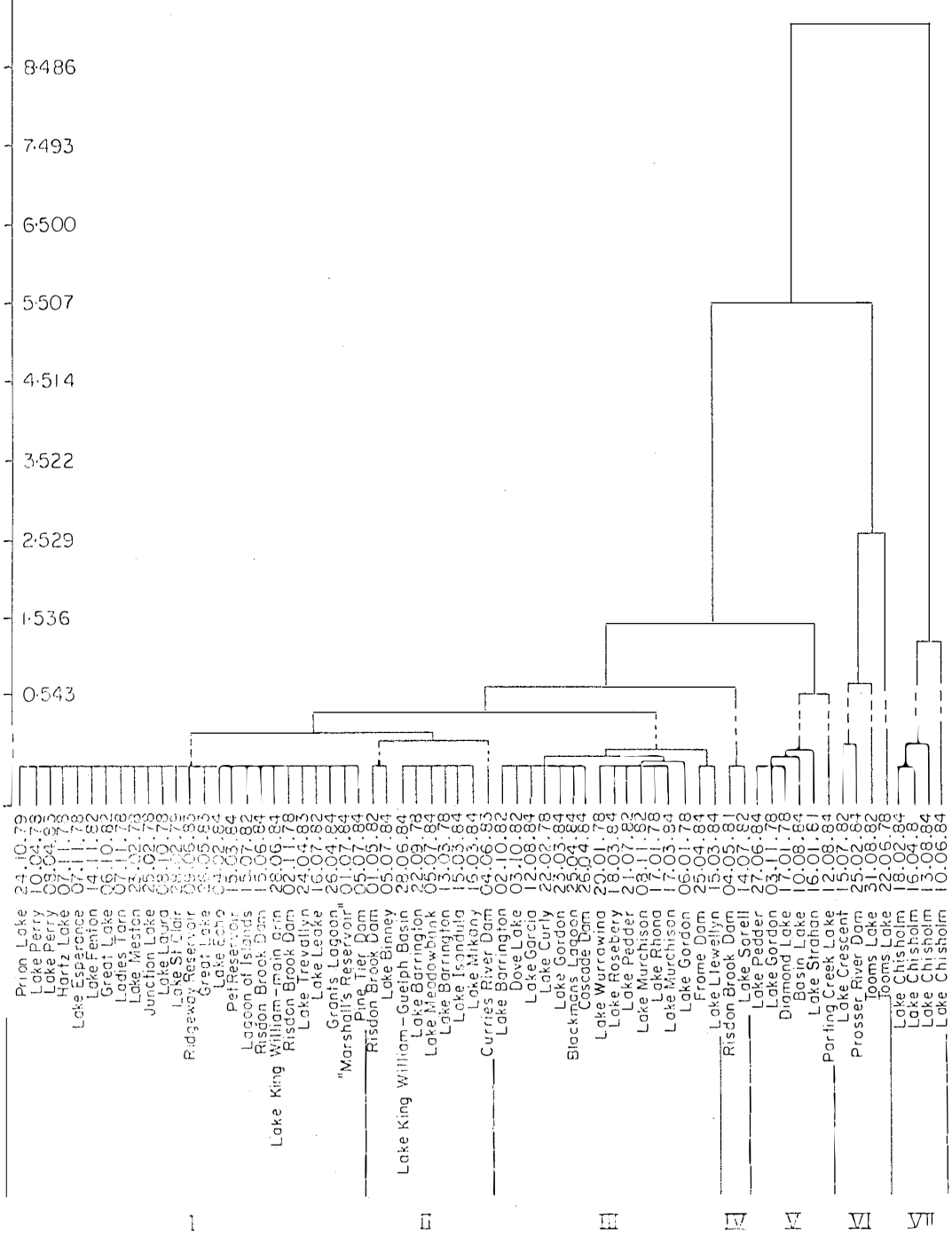


Figure 3.16 Dendrogram showing the arrangement of Tasmania's lakes into seven groups, based on their gilvin and turbidity values.

western areas (Province I of Buckney and Tyler) are moderately to highly dystrophic (Groups III and V, above), lying on ancient quartzitic rocks supporting highly humic vegetation, such as *Gymnoschoenus sphaerocephalus* (R.Br.) Hook. sedgelands, cool temperate rainforests, or wet heaths, and with an ionic character close to seawater. In comparison, lakes on Jurassic dolerite, such as those in the Central Plateau, Mt. Field, and the Hartz Range (mainly lying within Province II of Buckney and Tyler), whose ionic character lies between seawater and World Average Freshwater, are the clear, non-turbid types of Group I. The few turbid lakes, forming two discrete groups (IV, VI), lie in the dryer, rainshadow areas of central and eastern Tasmania (mainly Province IV of Buckney and Tyler), where evaporation may exceed precipitation. Here the climate is closest to that of south-eastern Australia, where many similar turbid lakes occur (see Section 2.5.5). Group II includes lakes which are located either in areas between the two major rock types, or between the dryer and wetter regions of the south-east, and some come from the north coast (Province III of Buckney and Tyler). These lakes may therefore represent a transitional group between clear lakes and either the humic or turbid ones found in these areas.

The groups of the cluster analyses intergrade to a certain extent, and the several categories of dystrophic groupings would, by more subjective means, be simply grouped all together as "dystrophic". The presence of some lakes, sampled on several occasions, in more than one category of the dendrogram (e.g. Lake Barrington, Lake Gordon), is an indication of the extent of temporal change. The most variable of the study sites, Risdon Brook Dam, thus fell into three groups of the classification, due to its differing colour and turbidity at different sampling dates. By the same token, the edaphic differences of the catchments of the Main Arm (clear waters) and the Guelph Arm (dystrophic) of Lake King William were recognized by the cluster analysis. Polyhumic Lake Chisholm was also separated from the other humic lakes as Group VII , on account of its much higher humic content.

The classification reveals that Tasmanian lakes have distinctive underwater light climates, which are dependant on their gilvin and turbidity content. Thus, for many lakes, the general optical characteristics can be predicted, with a considerable measure of confidence, from laboratory measurements of gilvin, and perhaps

turbidity, alone. Estimates of mean downwelling attenuation coefficients can be calculated from such measurements by use of the empirical relationships devised by the regression analyses. Likewise, laboratory measurements can be used to estimate the spectral distribution of PAR in lakes by allowing them to be placed into one of the categories of the classification, for which comparable spectroradiometric data is available from lakes which typify that group. Although it is not suggested that such methods permit the accurate prediction of quantum irradiance at any depth (Bowling and Tyler, 1986; Howard-Williams and Vincent, 1984; Jewson et al, 1984), for ecological or management purposes, prediction of optical water quality, such as attenuation coefficients and the spectral distribution of underwater light, is important (Kirk, 1982), especially when species composition and primary production are considered. The results of this survey permit these predictions to be made for Tasmanian lakes, where in situ measurement would be impossible, using only a small sample of water.

A similar optical classification was made on the standing freshwaters of north-east New South Wales (Figure 3.17). The majority of the lakes (25) all fell into one group (Group III), this representing those of low colour ($\bar{x}$ = 0.836 m^{-1}, σ = 0.645) and of slight turbidity ($\bar{x}$ = 1.93 N.T.U., σ = 1.11). The typical $Kd_{(av)}$ value for these waters would be around 0.97 m^{-1}, and their spectral distributions are represented by measurements from Lakes Hiawatha, Minnie Water, and Keepit, and from Chaffey, Storm King, Copeton, Malpas and Pindari Dams.

The other groups of the optical classification were all small. The moderately turbid and coloured Dumaresq and Oaky Dams, along with Yalgoo Reservoir made up Group I (Mean turbidity = 3.87±0.46 N.T.U.; mean g_{440} = 5.527±0.951 m^{-1}), while the more turbid ($\bar{x}$ = 6.53 N.T.U., σ = 0.74) but less coloured ($\bar{x}$ = 1.828 m^{-1}, σ = 1.150) Clarrie Hall and Quipolly Dams, Lake Inverell, and Beardy Waters Reservoir constitute Group II. Two dystrophic coastal lakes, Cooks Lagoon and Lake Ainsworth, made up a group of their own (Group IV), while highly coloured and increasing turbid St. Helena Dam, Kentucky Creek Dam, and Yarrie Lake, were each separated into groups of their own.

Overall, the results of this optical classification show that the majority of the lentic freshwaters of north-east New South Wales are optically homogeneous, and only slight differences exist between their underwater light climates. However, some exceptions to this general tendancy do exist, and differences of some individual study sites were

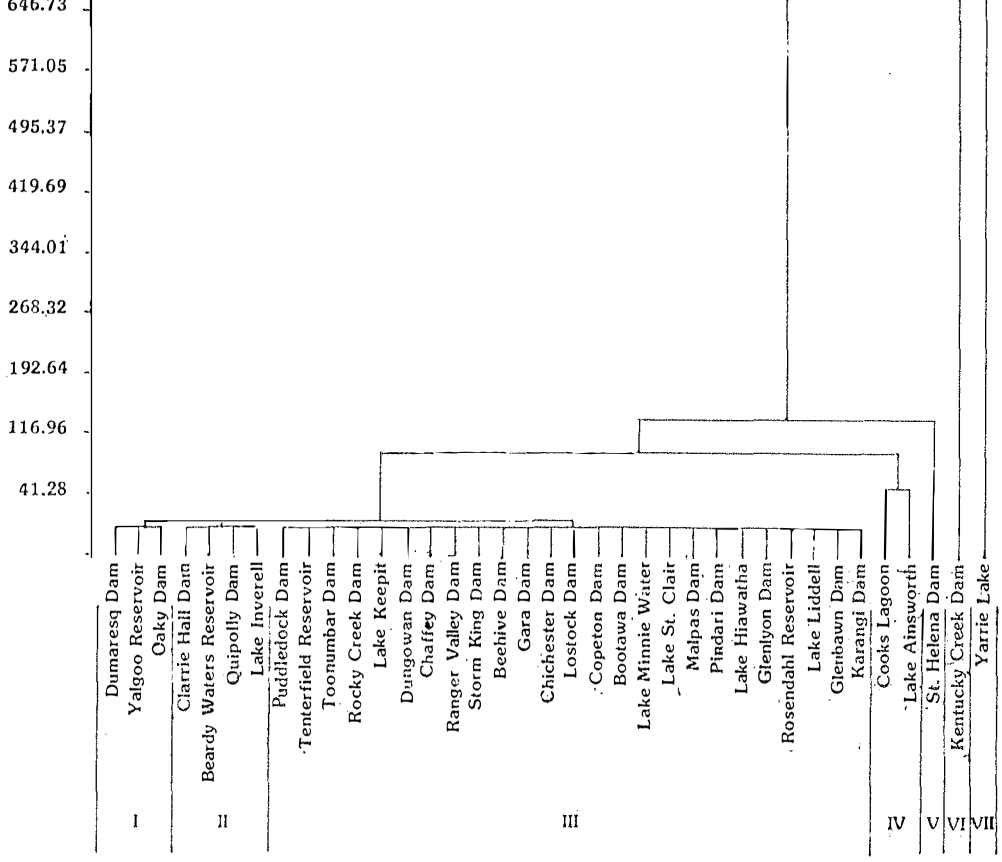


Figure 3.17 Dendrogram showing the arrangement of north-east New South Wales lentic freshwaters into seven groups based on their gilvin and turbidity values.

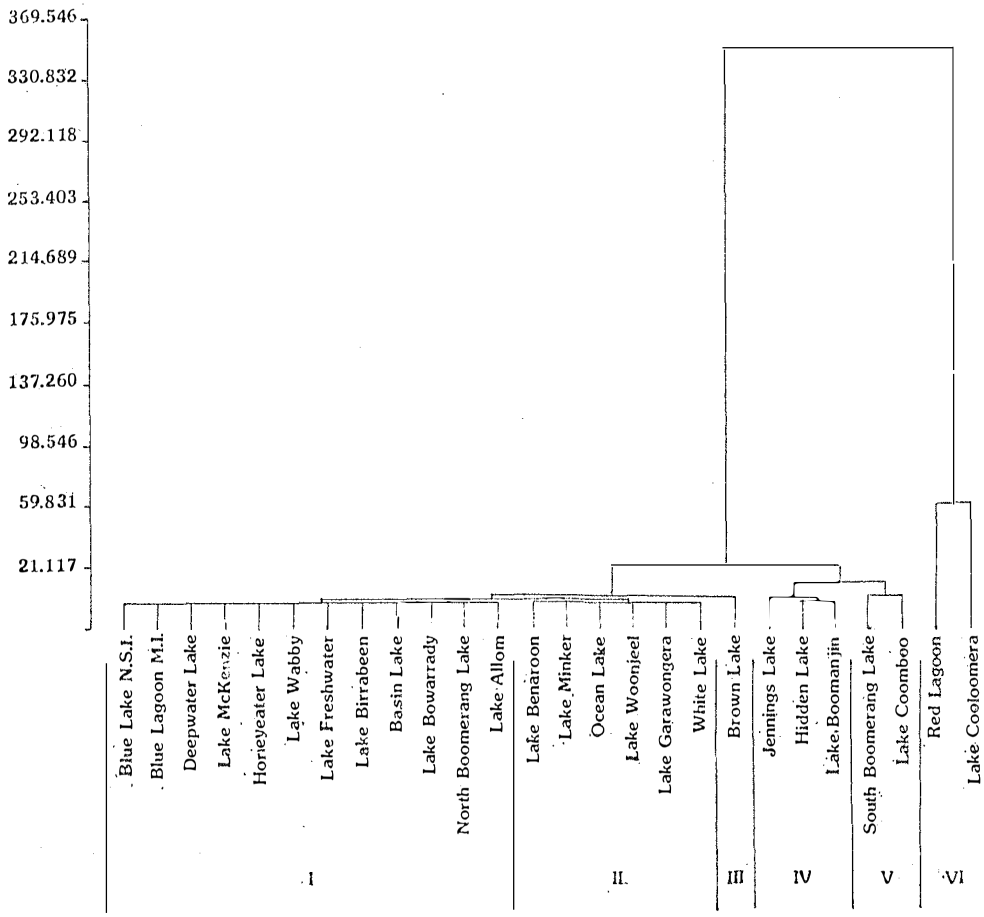


Figure 3.18 Dendrogram showing the arrangement of south-east Queensland coastal lakes into six groups based on their gilvin and turbidity values.

extreme. Unlike the optical classification of Tasmanian lakes, no geographical distributions of the lakes in the groups are obvious, and those that differ markedly from the typical optical properties of the area probably do so due to local effects.

Apart from possibly helping split Brown Lake off into a group of its own (Group III), turbidity differences between the lakes were too slight to contribute much to the optical classification of Queensland lakes (Figure 3.18). The remaining lakes can be grouped according to their gilvin concentrations alone. Group I includes all those with the clearest waters, with very low levels of g_{440} ($\bar{x}$ = 0.242 m^{-1}, σ = 0.300), while Group II includes those with only slight levels of colour ($\bar{x}$ = 1.180 m^{-1}, σ = 0.188). Three mesohumic lakes, Jennings, Hidden, and Boomanjin, form another distinct group (Group IV), while Lake Coomboo and the southernmost of the Boomerang Lakes combine to form Group V. These two groups have mean g_{440} values of 3.320±0.780 m^{-1}, and 8.550±1.181 m^{-1}, respectively. Finally, highly dystrophic Red Lagoon and Lake Cooloomera constitute Group VI.

This optical classification of Queensland freshwater coastal dune lakes, based on their humification, may result partly from their mode of origin. It has been suggested (Bayly, 1964; Bayly *et al*, 1975; Bensink and Burton, 1975; Timms, 1982, 1986a) that optically clear lakes are of the water-table window type, compared to perched lakes, which are humic. However, exceptions do occur, revealed by the presence of the two Boomerang Lakes, both perched lakes, in separate groupings of the classification, the northernmost being placed the clear-water lake type Group I, and the southernmost one in dystrophic Group V. Similarily, both Jennings Lake and Lake Birrabeen, two lakes in such close proximity and elevation to be almost linked, also fall within different groups of the classification. Unfortunately the geomorphic origins of many of these lakes are poorly documented, and so cannot be stated for many in each grouping of this classification.

As well as cluster analyses, regression analyses were also undertaken to derive statistical relationships between the vertical attenuation coefficient, $Kd_{(av)}$, and the various attenuating components of the aquatic medium. These were done for each area, separately. However, some of the resulting relationships appear very similar, and comparisons using the t-test showed no significant differences (at the 5% level) between the slopes of the $Kd_{(av)}$ verses g_{440}

regressions (before log normal transformation), indicating that increasing gilvin concentration increases the attenuation of light in a similar manner in the waters of all three study areas. Further *t*-tests on the intercepts of the three regressions found only one pair, those for New South Wales and Tasmania, to be significantly different ($t_{(2),(98)} = 3.62$, $P = 0.001$). Because of this, the Tasmanian and New South Wales equations cannot be used interchangeably to estimate $Kd_{(av)}$ from a g_{440} measurement, but the other equations may be substituted for each other, with only slight differences in the result obtained.

The slopes of the regressions between $Kd_{(av)}$ and turbidity, prior to log normal transformation, were also *t*-tested. There were no significant differences (at the 5% level) between those of Queensland and Tasmania, and between New South Wales and Tasmania, but one was shown between Queensland and New South Wales regressions ($t_{(2),(59)} = 2.32$, $0.05 < P < 0.02$). However, *t*-tests between the intercepts of the other equations were significant ($t_{(2),(98)} = 2.71$, $P \cong 0.01$ between New South Wales and Tasmania; and $t_{(2),(88)} = 3.15$, $0.01 < P < 0.001$ between Queensland and Tasmania). These regression equations are therefore not interchangeable, with the significant differences between them probably resulting from turbidity being a poor predictor of $Kd_{(av)}$, especially in Tasmanian waters and the south-east Queensland coastal lakes. Similarly, variations in the contribution of turbidity to attenuation in each of the three areas does not allow interchangeable use of the multiple regression equations of turbidity and gilvin against $Kd_{(av)}$.

3.4.5. Phytoplankton Diversity and Distribution

Few local algal floras have been described from Australia (Ling and Tyler, 1986). Although taxonomically incomplete, and subject to the limitations of single samples (which take no account of temporal changes in community composition) and the mesh size of the sampling net, the survey of the north-east New South Wales freshwaters reveals a diverse planktonic flora. Many sites had a considerable range of taxa, while in contrast, almost monospecific blooms of *Mallomonas* sp. and *Ceratium hirundinella* were present in Puddledock and Quipolly Dams, respectively. Cyanobacteria and diatoms also formed blooms in a number of the lakes and reservoirs, but green algae and dinoflagellates were particularly widespread. One unusual and uncommon dinoflagellate, *Prorocentrum*

playfairi, first recorded in New South Wales freshwaters by Playfair (1919) (as *Exuviella lima* (Ehr) Schutt.), and only recently reported again from Tasmanian coastal lakes (Croome and Tyler, 1987a), was also collected from two lakes and a reservoir in coastal parts of north-east New South Wales, in this study.

Between lake comparisons are however not often possible, as the samples were collected at different times of year, and some variation may therefore be due to seasonal differences. Factors such as the water temperature (dependent on location and sampling time), and more particularly the extent of eutrophication of each waterbody, would probably also contribute to differences in phytoplankton. The underwater light climates may also have an effect. For example, the low light levels in extremely turbid Yarrie Lake would almost certainly cause its depauperate phytoplankton, despite its very high nutrient concentrations. May and Powell (1986) found the presence of individual taxa in Chaffey Dam was related to changes in waterflow, depth, temperature, the availability of nutrients, and successional changes.

Chlorophyll a values indicate low trophic levels and sparse phytoplankton populations in most of the freshwater coastal dune lakes of south-east Queensland. Bayly *et al.* (1975) found that desmids, Chlorococcales, *Peridinium*, and *Dynobryon* were the predominant algae in Fraser Island lakes, as was the case in this study, although they listed a less diverse flora. Although limited to spot samples taken with a 20 µm plankton net over a three week period, no distribution patterns based on the optical properties of the lakes were apparent from this survey. It has been proposed (Kirk, 1976b, 1979, 1981a; Eloronta, 1978; Jeffery, 1980) that taxa with accessory photosynthetic pigments may be at an ecological advantage in some underwater light climates. In contrast, Arthington *et al.* (1986) have attributed faunal differences between lakes of Fraser Island, particularly in the composition of littoral invertebrates and fish, to differences in their geomorphic origin and resultant water colour.

It is acknowledged that pilot surveys such as these are unlikely to show any distinct differences between phytoplankton communities. Quantitative sampling and counting at a species level would be required to determine if variations do exist in phytoplankton composition between lakes of differing optical properties.

3.5 CONCLUSIONS

The results of these studies have shown that, with a few exceptions, dissolved humic substances play the major part in the extinction of light in the standing inland freshwaters of the three areas. This is especially so in Tasmanian lakes, and those of the coastal dune areas of south-east Queensland, which cover a wide range of concentrations of dissolved humic substances. Although most lakes and reservoirs of north-east New South Wales have only slight to moderate concentrations of gilvin, and only a few are highly dystrophic, it never-the-less still plays the dominant role in attenuating P.A.R. in these waters too. Most New South Wales waters are also slightly to moderately turbid, and a few are highly turbid, so suspended particulate matter also contributes to attenuation in these lakes and reservoirs, although usually to a much lesser extent than gilvin. In comparison, and unlike other inland waters from other parts of Australia, lakes from Tasmania and south-east Queensland are usually non-turbid, and tripton is an important attenuator in only a very few of them. However, even in turbid waters, humic materials also contribute greatly to light extinction, due to their sorption onto the outside of suspended mineral particles (Kirk, 1980a). Chlorophyll a is rarely a consideration, except in some of the more eutrophic lakes, such as those of north-east New South Wales, and water itself would be a significant attenuator of P.A.R. only in some of the clearest lakes of the three areas, in particular those from Tasmania and Queensland.

A predictive optical classification was devised for Tasmanian lakes which closely follows one based on chemical and edaphic features (Buckney and Tyler, 1973a). A similar classification of north-east New South Wales waters was less successful, due to the majority of these waters being optically fairly homogeneous, but one for the coastal dune lakes of south-east Queensland showed distinct groups differing mainly in gilvin concentration. Statistical relationships were derived between $Kd_{(av)}$ and the various attenuating components of the aquatic medium for each of the three areas, with those for gilvin being very similar, indicating that gilvin attenuates light in a nearly identical manner in the waters of all study areas.

Some temporal variation in humic content was evident in the Queensland lakes, as their transparency was better than in earlier studies (e.g. Bayly *et al.*, 1975; Arthington *et al.*, 1986), implying lower gilvin concentrations and higher pH values. The humic nature of

of these lakes was the cause of their acidity, with a strong negative correlation apparent between pH and gilvin concentration, confirming similar observations by Bayly (1964) and Bayly *et al*., (1975). In contrast, levels of humics were insufficient to affect pH in the New South Wales waters, and their presence was offset by buffering by the dominant anion, bicarbonate, and salinities of up to 300 mg L^{-1} (Timms, 1970; Banens, in press).

Diverse phytoplankton floras were found in both the New South Wales and Queensland freshwaters. Species from the Division Chlorophyceae, in particular desmids and Chlorococcales, were abundant, although flagellates and diatoms were also present in considerable numbers, and cyanobacteria were common in some New South Wales waters. However the data are insufficient to show whether various ecological factors, and especially the optical properties of the various waters, are influencing the distributional patterns of these algae. Investigations using quantitative techniques would be required to determine if correlations exist between species composition of phytoplankton communities, and the humic content of lakes. However, the surveys do provide additional information to the depauperate knowledge of the phytoplankton of these two regions, providing a basis for future research on this topic.

CHAPTER FOUR

PHYSICO-CHEMICAL STUDIES OF FRESHWATER COASTAL LAGOONS FROM WESTERN AND SOUTH-WEST TASMANIA, AND FROM KING AND FLINDERS ISLAND, BASS STRAIT

4.1 INTRODUCTION

4.1.1 The Aims and Scope of this Study

Freshwater coastal dune lakes occur in numerous locations along the Australian and Tasmanian coastlines, and studies of these have been reviewed in Section 2.3. Although originating from a number of geomorphic processes, they are typically acidic and of low salinity, with an ionic chemistry dominanted by sodium and chloride ions. Water clarity varies considerably, with some being clearwater lakes, while others are highly dystrophic (Timms, 1982, 1986a - see also Chapters Two and Three).

The Bass Strait islands have many lentic waterbodies, and others are located along the west and south-west coasts of Tasmania. Despite the mainland Australian coastal lakes having been considerably studied, those of Tasmania have attracted little limnological attention. The majority are highly dystrophic, and thus are ideal sites for studies of humic waters. Additionally, humic coastal lagoons of Tasmania have very rich planktonic floras (Croome and Tyler, 1987a,b; Croome et al, 1987 and in press), and those of the Bass Strait islands are likely to, too. Because of the emerging phycological importance of these waters, this study aimed at investigating their physico-chemical properties. The extent of any limnological similarities or differences between King Island lagoons, due to local variations in dune geochemistry, formed a suitable subject for multivariate analysis, and the lagoons of both Bass Strait islands were compared in the same manner. The remoteness of many prevented the direct measurement of the spectral distribution and attenuation of light within them,

but it is possible to speculate on their underwater light climates, using the proce dures developed in Chapter Three of this thesis. The species distribution and diversity of the phytoplankton of some of the lagoons of western and south-western Tasmania, and the influence of humics on this, were also examined.

4.2 METHODS

4.2.1 Collection and Analysis of Samples

Surface samples from six Flinders Island and seventeen King Island lagoons were collected in May, 1982, after an extremely dry summer. A sample of groundwater from a freshwater spring near Surprise Bay, King Island, was also collected. These sites, shown in Figure 4.1 and identified in Table 4.1, represent only a small proportion of the standing waters on the two islands. The samples, in one litre black polyethylene bottles, were airfreighted to Hobart and stored at 4°C pending laboratory analyses.

Most samples from western and south-western Tasmania were collected in January and February, 1987, and include some obtained by P.A. Tyler, R.L. Croome, and B.V. Timms from the South-east Cape and New River Lagoon areas, and from Hibbs Lagoon. Rebecca Lagoon was sampled in February, 1984, and additional data from November, 1984, is included for Lakes Garcia, Strahan, and Ashwood. The sampling locations are shown in Figure 4.2, and identified in Table 4.2. Access to many of the remoter sites was by helicopter, which limited collections to one litre water samples, taken in black polyethylene bottles, and to phytoplankton, obtained by tows with a 20 μm plankton net. All water samples were stored at 4°C, and the ionic analyses were done by the Tasmanian Government Analyst. Plankton samples were fixed with 10% formalin for microscopic identification using the texts referenced in Section 3.2.2.

Water colour was determined as g_{440} (Kirk, 1976b), and (for King Island) on the Hazen (mg Pt L^{-1}) scale (see Section 3.2.2 for details of these methods). Complete absorption scans (400-750 nm) were undertaken on selected filtered (0.45 μm) King Island samples against distilled water blanks, using a Pye Unicam SP 8/100 U.V./VIS. double beam spectrophometer. The turbidity of the samples was measured using a Hach 2100 turbidimeter, against formazan standards.

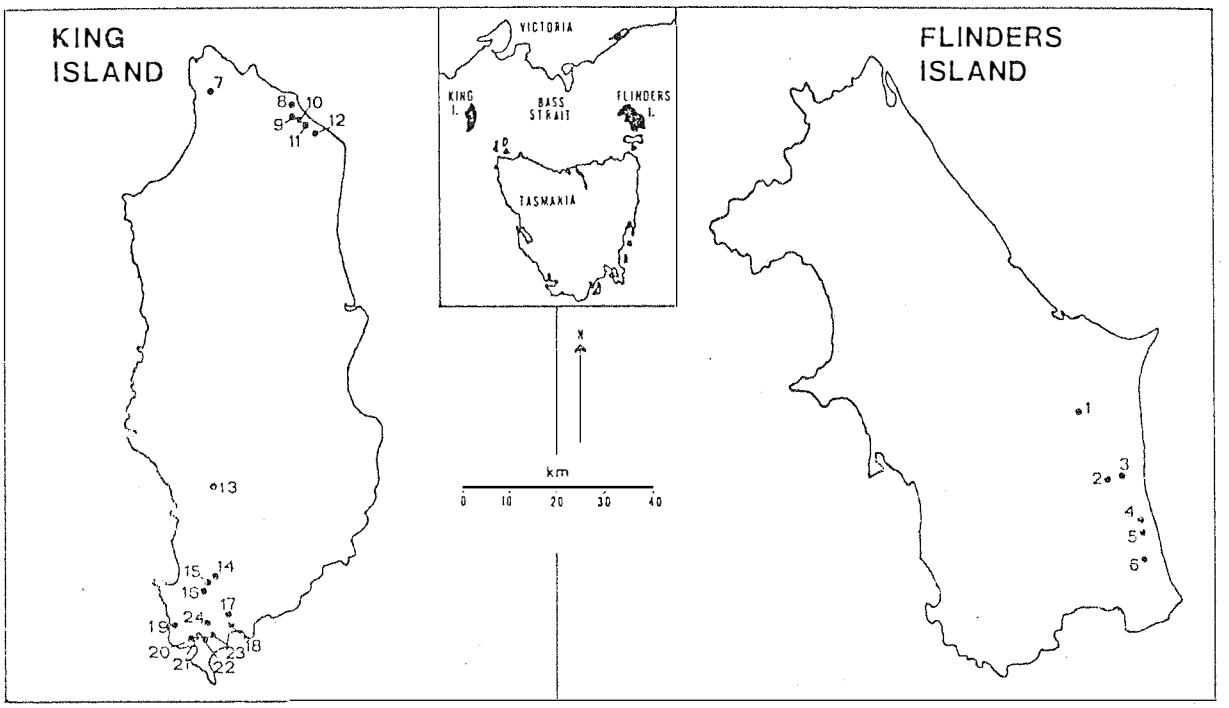


Figure 4.1 Location of sampling sites of coastal lagoons on King and Flinders Islands, Bass Strait. The numbers refer to the lagoons listed in Table 4.1.

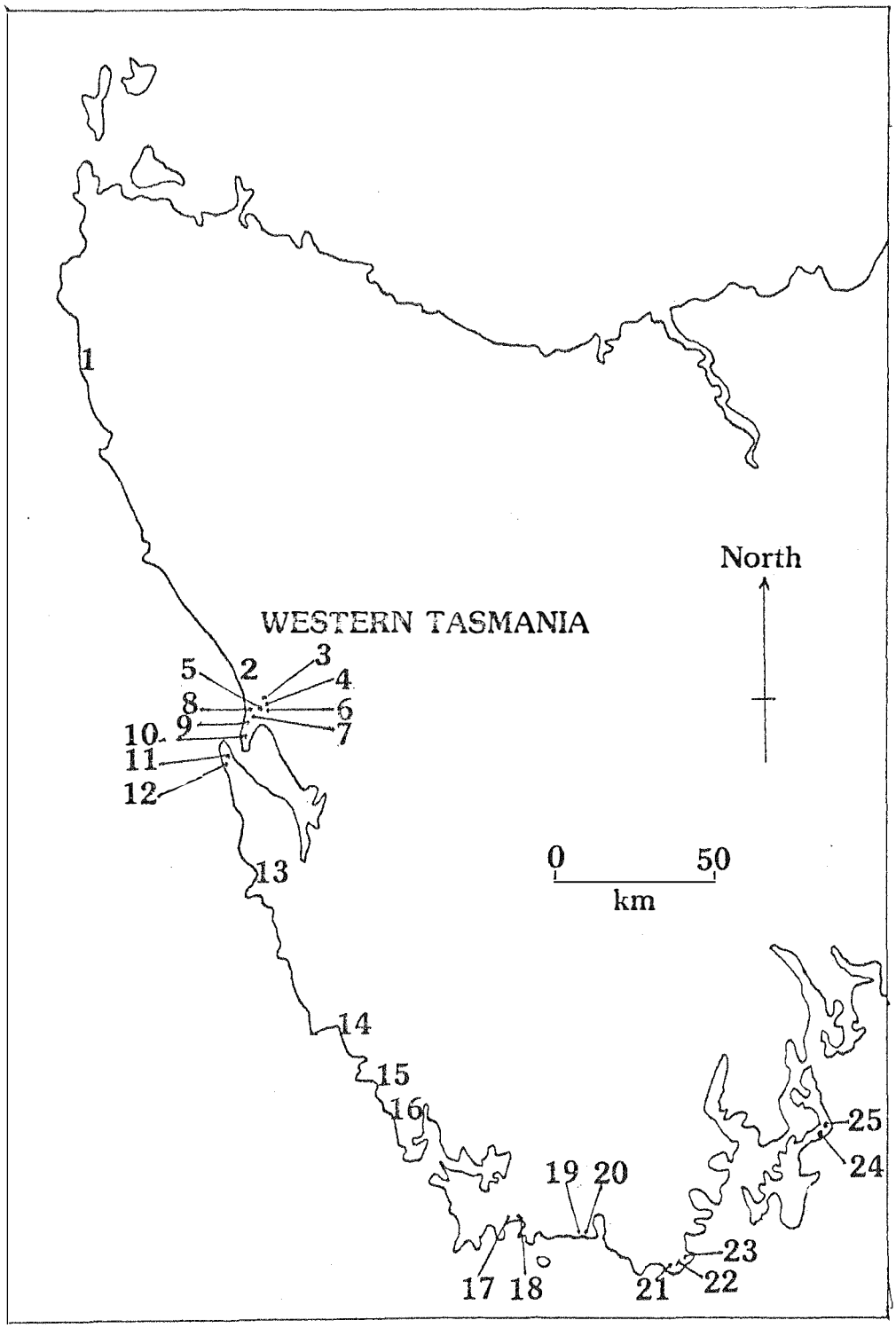


Figure 4.2 Location of sampling sites of coastal lagoons from western and south-west Tasmania. The numbers refer to the lagoons listed in Table 4.2.

pH and electrical conductivity at 18°C (K_{18}) were measured electrometrically in the laboratory using Radiometer equipment. Chemical analyses were performed on filtered (0.45 µm) samples, with suitable dilutions when necessary. Sodium, potassium, and magnesium ion concentrations were determined by atomic absorption spectroscopy, and calcium by the colourimetric method of Kerr (1960). Bicarbonate was determined by potentiometric titration against 0.01N HCl to pH 4.5 (Golterman et al, 1978), chloride by conductimetric titration against $AgNO_3$ (Golterman et al, 1978), and sulphate turbidimetrically using barium chloride (A.P.H.A., 1971). Salinity, in mg L^{-1}, was calculated as the sum of these seven major ions. Total dissolved iron was analysed by atomic absorption spectrophometry, while silica was measured by the molybdate yellow method (A.P.H.A., 1971). Total phosphorus was determined following predigestion with potassium persulphate by the stannous chloride method for the Bass Strait island samples, but the ascorbic acid method was used for samples from western and south-western Tasmania. Total nitrogen was analysed following D'Elia et al (1977). A Technicon Autoanalyser II was used for the nutrient analyses of the samples from south-west Tasmania, which were collected in acid washed plastic bottles.

The lakes near Strahan are more easily accessable, and were sampled from an inflatable rubber boat. Thermal and conductivity profiles were measured with a W.T.W. LF 191 conductivity meter, and dissolved oxygen with a W.T.W. OXI 91 oxymeter. pH was also measured in situ with a Methrom E558 pH meter, and water transparency with a 20 cm quartered black and white Secchi disc.

A principle co-ordinates analysis (Gower, 1966), using fourteen variables - turbidity, g_{440}, pH, K_{18}, dissolved iron, SiO_2, total P, the four cations (expressed as a percentage of total cations), and the three anions (expressed as a percentage of total anions) - was employed to show variability between the twenty-four samples from the Bass Strait islands. Pearson product-moment correlations (r) were calculated between parameters using the SPSS computer program (Nie et al, 1975).

4.3 RESULTS

4.3.1 Thermal and Oxygen Profiles

Thermal and oxygen profiles from eight lakes near Strahan, and from Hibbs Lagoon, are shown in Figure 4.3. Apart from clearwater Lake Bantic, and extremely shallow South Strahan Lagoon, all showed a marked thermal gradient with depth. Little Bellinger Lake showed a temperature drop of 5.5°C over just three metres, while there were similar changes over 1.3 metres depth in Lake Strahan, and 4.0 metres in Hibbs Lagoon. Only Lake Garcia displayed a thermal profile typical of a thermally stratified lake, with a distinct epilimnion, a thermocline between five and six metres deep, and a colder hypolimnion.

The oxygen profile confirmed the existance of thermal stratification in Lake Garcia. Dissolved oxygen decreased markedly at the same depth as the thermocline, to hypolimnetic levels of less than 10% saturation. A marked oxycline was also present in Little Bellinger Lake, changing from almost complete saturation at the surface to nearly anoxic conditions close to the sediments, despite the shallowness of this lake. Oxygen also decreased slightly with depth in Lake Koonya, especially over the bottom metre, but oxygen profiles in this and other lakes indicate frequent mixing and ventilation throughout.

4.3.2 Turbidity and Colour

Results from the Bass Strait islands are listed in Table 4.1. Turbidity values were generally below 1 N.T.U., although three King Island lagoons, Shearing Shed, Ridge, and Dead Sea, were markedly turbid, with Dead Sea much more so than previously recorded (Buckney and Tyler, 1976). Turbidity was measured for only four of the lagoons from western and south-west Tasmania, and was below 1.0 N.T.U. in all cases. Qualitative observations indicate that most of these coastal lagoons would have low turbidity.

The most notable feature of many of these waters is their extremely high gilvin concentrations. The King Island lagoons all contained some dissolved organic colour. Colour was least in Penny Lagoon (g_{440} = 3.455 m^{-1}), but most lagoons were considerably more dystrophic, and four had extreme amounts of organic colour. In contrast, the six Flinders Island lagoons were all virtually uncoloured, with g440 levels below 2.0 m^{-1} (Table 4.1). Median water colour for the two islands was 6.736 m^{-1}.

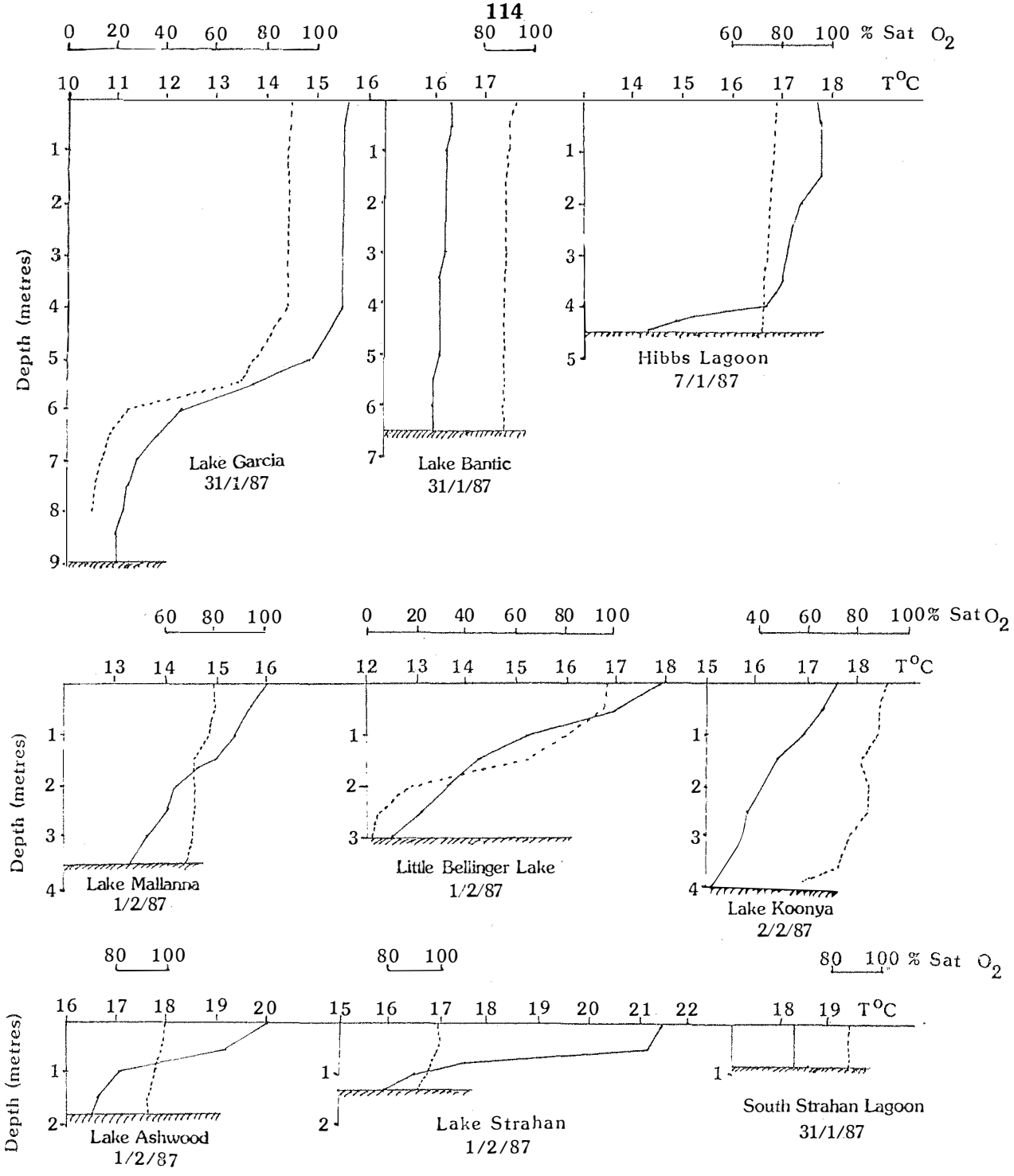


Figure 4.3 Summer thermal (——) and oxygen (- - -) profiles from lagoons from the west coast of Tasmania.

Table 4.1: Physicochemical characteristics of lagoon waters (in order of decreasing salinity) from King and Flinders Islands Blank space indicates no analysis was conducted. 0 indicates the characteristic was undetectable. Grid references are given for lagoons that are un-named.

Sample Site and grid reference	Sample No.	Turbidity (NTU)	Gilvin (g_{440}, m^{-1})	Gilvin (Pt units, mg L^{-1})	pH	K_{18} ($\mu S\ cm^{-1}$)	Na^+	K^+	Ca^{2+}	Mg^{2+}	Cl^-	SO_4^{2-}	HCO_3^-	Salinity (mg L^{-1})	Fe (mg L^{-1})	SiO_2 (mg L^{-1})	Total P ($\mu g\ L^{-1}$)	$\frac{\Sigma^-}{\Sigma^+}$ (%)
							meq L^{-1}											
Flinders Island																		
South Chain Lagoon	5	0.50	1.516		8.00	50380	672	17.1	68.8	173	740	56.3	3.95	48804	0.50	0.0	40	86
Logans Lagoon	6	0.50	1.727		7.20	36660	435	13.3	51.6	142	485	100	3.64	35672	0.40	1.2	90	92
Sandy Lagoon	2	0.50	1.670		7.00	30150	333	11.0	57.5	82.2	400	61.5	2.71	27546	0.30	8.0	40	96
North Chain Lagoon	4	0.50	1.612		7.80	23800	228	7.62	57.5	69.9	295	43.8	3.08	20309	0.30	2.1	50	94
No Duck Lagoon	3	0.50	0.345		7.30	17100	130	5.13	74.6	34.5	194	50.0	1.06	14464	0.25	0.3	40	100
Sticks Lagoon	1	0.50	0.633		4.90	681	4.96	0.217	0.438	1.23	5.70	0.052	0.034	353	0.90	4.0	115	85
King Island																		
Big Lake	18	0.70	7.082	90	8.15	12782	111	3.99	6.00	60.2	148	15.8	3.33	9775	0.00	0.3	90	92
Woodlands Lagoon	23	0.65	6.506	80	7.80	2374	15.1	0.537	4.13	7.40	17.1	5.73	3.79	1655	0.00	0.7	65	98
Lake Flanigan	7	0.50	7.370	100	8.30	2024	14.2	0.588	2.35	6.85	15.6	0.969	4.43	1352	0.10	0.1	20	88
Seal Rocks Lagoon	19	0.80	13.876	175	7.00	2292	16.6	0.775	1.05	6.95	20.2	0.990	1.28	1347	0.20	0.1	160	92
Dembys Lagoon	20	2.10	23.836	300	7.30	2071	14.4	0.430	2.10	5.65	17.2	2.66	1.15	1274	0.40	0.2	60	93
Pioneer Lagoon	22	0.90	9.788	125	7.90	1965	17.4	0.512	2.85	4.61	15.8	1.66	3.50	1196	0.00	0.1	50	82
Shearing Shed Lagoon	21	19.00	18.539	200	6.50	1870	12.8	0.476	0.900	4.34	15.1	1.02	2.31	1109	0.20	0.1	350	99
Attrills Lagoon	16	1.30	5.930	70	7.95	1330	7.87	0.192	5.17	3.91	7.60	3.75	2.91	967	0.20	1.1	150	83
Pearshape Lagoon	15	1.90	6.851	80	7.40	1310	7.57	0.338	2.78	3.55	9.70	0.198	3.70	865	0.25	0.1	570	96
Granite Lagoon	8	0.50	8.348	110	7.30	1254	10.3	0.384	1.33	3.17	10.75	0.917	0.557	773	0.00	0.0	0	81
Penny Lagoon	12	0.25	3.455	20	7.40	909	7.83	0.384	0.575	2.14	7.95	0.313	0.771	577	0.00	0.0	30	83
Sullivans Lagoon	14	0.80	6.333	80	7.80	776	3.91	0.090	2.08	2.30	4.75	1.00	2.52	534	0.10	1.2	20	88
Groundwater near Surprise Bay	24	1.00	1.727	20	7.55	697	2.17	0.205	2.31	1.89	1.10	2.16	3.05	456	0.10	0.4	60	96
Lake Martha Lavinia	11	1.00	35.236	450	5.55	693	4.13	0.384	0.575	1.89	5.85	0.263	0.115	372	0.30	0.5	230	89
Lagoon north of Big Lake 54 GYA 388573	17	1.00	52.796	600	4.90	626	4.39	0.038	0.400	1.65	5.30	0.183	0.082	332	0.70	0.1	290	86
Un-named Lagoon 55 GBS 472078	9	0.55	38.115	500	4.25	596	4.17	0.230	0.425	1.32	4.95	0.252	0.000	317	0.20	1.8	30	85
Dead Sea	13	29.00	6.621	30	5.40	545	3.94	0.090	0.275	1.30	4.24	0.229	0.082	283	0.85	1.0	400	81
Ridge Lagoon	10	15.00	47.787	600	4.40	353	2.47	0.160	0.325	0.847	3.08	0.060	0.000	192	0.20	3.1	110	83
MEDIAN		0.75	6.736	105	7.30	1600	11.6	0.407	2.21	4.13	12.9	1.01	2.42	1038	0.20	0.35	63	89
MEAN		3.33	12.821	202	6.88	8051	86.0	2.67	14.4	25.9	101	14.6	2.00	7105	0.27	1.10	128	89
STANDARD DEVIATION		7.19	15.364	199	1.26	13663	169	4.76	25.3	47.0	192	26.9	1.55	13077	0.25	1.80	143	6

Table 4.2: Optical, chemical, and nutrient parameters from the coastal lakes and lagoons of western and south-west Tasmania

Lake Number	Name	Date Sampled	g_{440}	Secchi Depth	Field pH	Lab pH	K_{18}	Na^+	K^+	Ca^{2+}	Mg^{2+}	Cl^-	SO_4^{2-}	HCO_3^-	$\frac{\Sigma^-}{\Sigma^+}$	Salinity	SiO_2	Total N	Total P
			m^{-1}	m			μScm^{-1}	← meq L^{-1} →							%	← mg L^{-1} →		← μg L^{-1} →	
1	Rebecca Lagoon	11-2-84	3.397	-	-	8.15	1192	10.865	0.175	0.865	3.570	8.450	1.050	3.736	85.5	895.6	-	-	-
2	Lake Mallana	1-2-87	27.233	0.70	4.25	4.20	128	0.757	0.031	0.060	0.207	0.893	0.044	0.000	88.8	56.1	4.18	470	10
3	Lake Bantic	31-1-87	0.173	> 6.50	5.60	5.70	103	0.713	0.023	0.060	0.165	0.811	0.001	0.025	87.1	50.9	0.14	320	4
4a	Lake Garcia	9-11-84	9.326	-	-	4.90	105	0.934	0.015	0.028	0.196	0.830	0.060	0.033	78.7	61.1	-	-	-
4b		31-1-87	6.448	2.00	5.10	5.00	105	0.696	0.015	0.060	0.165	0.831	0.013	0.008	91.0	50.9	0.57	160	8
5a	Lake Ashwood	9-11-84	11.052	-	-	4.60	152	0.808	0.030	0.057	0.343	1.080	0.303	0.014	112.8	78.8	-	-	-
5b		1-2-87	4.894	>1.80	4.70	4.60	140	0.843	0.021	0.090	0.240	1.000	0.115	0.002	93.5	66.9	1.90	280	6
6	Little Bellinger Lake	1-2-87	12.667	1.00	4.60	4.60	110	0.696	0.015	0.075	0.174	0.845	0.001	0.002	88.3	51.3	3.42	270	11
7	Lake Koonya	2-2-87	31.839	0.70	-	4.20	115	0.678	0.026	0.055	0.174	0.811	0.001	0.000	87.0	48.7	2.23	360	13
8	Ocean Beach Lagoon	19-2-87	13.933	-	-	4.70	550	4.130	0.072	0.190	1.099	4.930	0.508	0.004	99.1	315.3	< 0.10	450	25
9a	Lake Strahan	9-11-84	28.552	-	-	4.75	225	1.660	0.042	0.013	0.469	1.860	0.274	0.030	99.1	126.8	-	-	-
9b		1-2-87	27.406	0.60	4.65	4.60	233	1.652	0.038	0.080	0.421	1.887	0.090	0.002	90.3	118.5	1.26	520	58
10	South Strahan Lagoon	31-1-87	51.127	0.40	4.00	4.00	256	1.739	0.031	0.120	0.413	2.056	0.046	0.000	91.2	123.8	1.75	770	16
11	Heron Pond	18-2-87	45.369	-	-	4.00	295	2.000	0.041	0.140	0.446	2.423	0.063	0.000	94.6	144.8	1.28	550	10
12	Teal Pond	18-2-87	31.666	-	-	4.10	450	3.217	0.059	0.155	0.686	3.803	0.188	0.000	96.9	231.7	< 0.10	520	19
13	Hibbs Lagoon	7-1-87	8.579	-	6.80	6.30	145	0.870	0.036	0.140	0.322	0.930	0.100	0.184	88.7	77.1	-	-	-
14	Nye Bay Lagoon	16-2-87	3.282	-	-	7.20	480	2.957	0.079	1.650	0.793	3.493	0.346	0.574	80.5	289.3	0.11	330	28
15	Mulcahy Bay Lagoon	12-2-87	20.151	-	-	4.40	154	0.939	0.077	0.120	0.256	1.121	0.096	0.000	87.4	74.5	1.18	260	5
16	Paradise Lagoon	12-2-87	19.057	-	-	7.10	350	3.435	0.062	1.650	0.488	2.299	0.200	0.492	53.1	241.5	0.59	290	25
17	Freney Lagoon	13-2-87	8.751	-	-	4.00	92	0.591	0.026	0.055	0.116	0.648	0.001	0.000	82.4	40.2	0.48	100	8
18	Millar Lagoon	13-2-87	10.018	-	-	4.30	308	0.635	0.023	0.055	0.149	0.797	0.004	0.000	92.9	46.9	0.65	200	9
19	Pond 1 New River	15-1-87	5.988	-	-	4.80	120	0.774	0.038	0.060	0.182	0.930	0.156	0.004	103.4	64.2	< 0.10	-	-
20	Pond 2 New River	15-1-87	8.636	-	-	4.80	130	0.826	0.046	0.070	0.192	1.000	0.156	0.005	102.6	68.5	< 0.10	-	-
21	S.E. Cape Lagoon 1	8-9-85	-	-	-	5.20	366	2.652	0.049	0.135	0.612	2.901	0.292	0.026	93.4	191.6	-	-	-
22	S.E. Cape Lagoon 2	8-9-85	-	-	-	4.30	280	1.957	0.041	0.075	0.405	2.141	0.229	0.000	95.6	140.0	-	-	-
23	S.E. Cape Lagoon 3	8-9-85	-	-	-	4.30	294	2.130	0.044	0.075	0.421	2.256	0.250	0.000	93.9	144.3	-	-	-
24	Gibbs Lagoon	23-1-87	253.33	-	-	4.60	1510	13.174	0.336	0.740	2.645	15.042	1.417	0.030	97.6	966.7	19.8	-	-
25	Big Lagoon	23-1-87	14.854	1.25	7.10	6.50	770	5.565	0.162	0.400	1.388	6.507	0.563	0.133	95.9	425.2	0.12	-	-
	MEDIAN		12.667	1.00	4.70	4.60	229	1.300	0.040	0.078	0.374	1.491	0.136	0.004	92.1	98.7	0.62	325	11
	MEAN		16.850*	-	5.20	5.00	327	2.425	0.059	0.260	0.598	2.592	0.235	0.189	91.1	185.4	1.99	366	16.4
	STANDARD DEVIATION		13.489*	-	1.10	1.08	334	2.995	0.066	0.440	0.778	3.072	0.323	0.709	10.4	231.4	4.35	171	13.4

* Excludes g_{440} data from Gibbs Lagoon

Lake Bantic was the only non-dystrophic coastal lake from western and south-west Tasmania, while Gibbs Lagoon was the most highly coloured of all, with its shallow waters being almost black due to its g_{440} of 253 m^{-1}. The median value for this area was 12.67 m^{-1}. Some temporal variation is apparent in Lake Garcia and especially Lake Ashwood (Table 4.2).

Absorption spectra of filtered water from a number of King Island lagoons, ranging from the most dystrophic to the least dystrophic, are shown in Figure 4.4. These vividly reveal how light attenuation increases, especially in the shorter wavelengths, as gilvin levels rise, although absorption of longer wavelength light is also increased. The scan for Dembys Lagoon (number 20) is comparable to that of Sulphide Pool, the most dystrophic of the Gordon River lakes (King and Tyler, 1982a, Figure 26).

4.3.3 Secchi Disc Depth

The few Secchi disc measurements for the lakes near Strahan (Table 4.2) indicate how rapidly light is extinguished with depth in the highly dystrophic waters of some coastal lakes. Water transparency was moderate in mesohumic Lakes Ashwood and Garcia, while the greatest clarity was observed in Lake Bantic, where the Secchi disc was still visable on the bottom, at 6.50 metres.

4.3.4 pH, Conductivity and Salinity

The lagoons of the Bass Strait islands showed a considerable range of pH values, although most were close to or above neutral (Table 4.1). Sticks Lagoon was the only exception on Flinders Island, while there were six acidic lagoons on King Island, five of which had salinities of less than 375 mg L^{-1}. The higher pH values of other lagoons are probably maintained by their higher salinities, with concomitantly higher concentrations of bicarbonate providing buffering even in the presence of considerable amounts of yellow organic substances. Median pH for the two islands was 7.30.

In contrast, almost all the lakes from western and south-west Tasmania were acidic (Table 4.2), with the lowest pH, 4.0, being recorded for Heron Pond, and in South Strahan and Freney Lagoons. Only five had pH values greater than 6.0, and three were above neutral. The highest pH, 8.15, was measured in Rebecca Lagoon. Median pH for the area was 4.60.

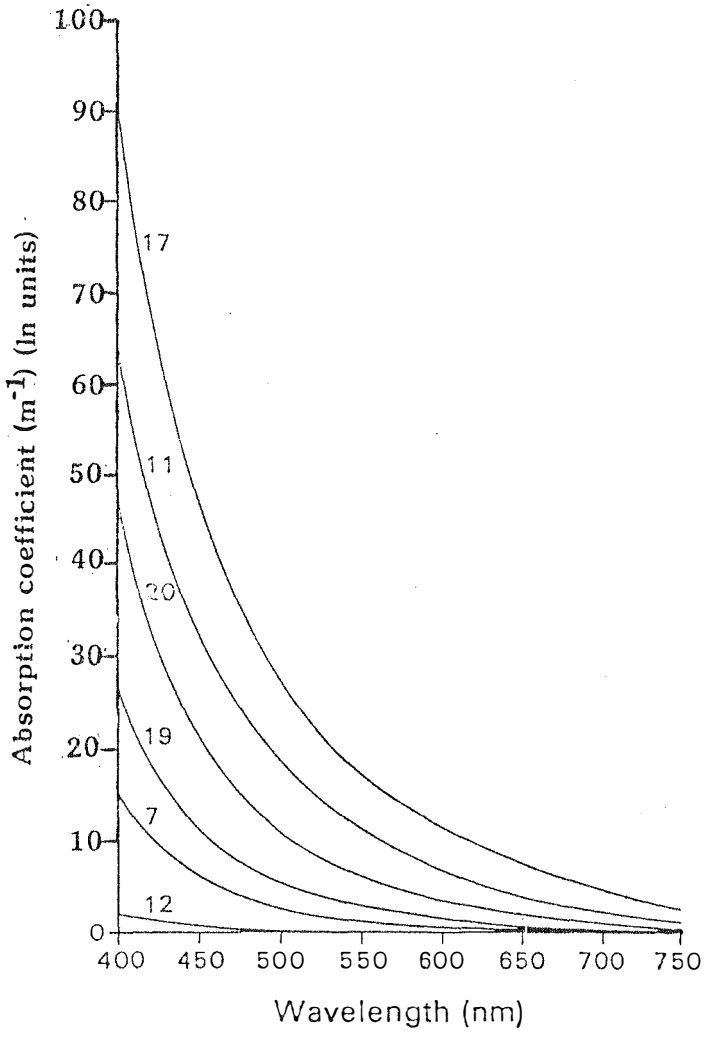


Figure 4.4 Absorption spectra of filtered water from selected lagoons on King Island, relative to distilled water. Numbers refer to the lagoons listed in Table 4.1.

Comparisons of measurements of pH made *in situ* in some western Tasmanian lagoons, and those made later on water samples in the laboratory, show only minor differences, never more than 0.10 of a unit. The laboratory measurements are therefore considered an acceptable determination of hydrogen ion concentration.

With the exception of Sticks Lagoon, the Flinders Island lagoons were considerably more saline than those of King Island (Table 4.1). South Chain Lagoon was hypersaline while Logans Lagoon had a salinity close to that of seawater. Using 3 g L^{-1} as the division between fresh and saline waters (Bayly and Williams, 1966), all the King Island lagoons apart from Big Lake can be classified as fresh, whereas Sticks Lagoon was the only fresh example from Flinders Island. Big Lake is subject to occasional incursions of seawater (Jennings, 1957), and the saline lagoons of Flinders Island are probably also receiving marine inflows. The median conductivity value for the Bass Strait islands was 1600 μS cm^{-1}.

The conductivities of most of the western and south-west Tasmanian coastal lagoons were low (Table 4.2). This is notable considering that most are within a kilometre from the sea, and all were fresh. Gibbs Lagoon and Rebecca Lagoon had the highest conductivities, with K_{18} values of 1510 μS cm^{-1} and 1192 μS cm^{-1} respectively, while the most dilute was Frenchy Lagoon, where K_{18} was 92 μS cm^{-1}.

Regression analysis showed that the salinities of the lagoon waters were closely related to their conductivities. The relationship calculated for the King and Flinders Island lagoons was:-

$\log_{10}$ Salinity = 1.12 $\log_{10}K_{18}$ - 0.575 (r^2 = .998, n=24, $P \ll 0.001$), while the regression equation for the western and south-west Tasmanian coastal lagoons was:-

$\log_{10}$ Salinity = 1.10 $\log_{10}K_{18}$ - 0.529 (r^2 = 0.914, n=28, $P \ll 0.001$)

Logarithmic transformations were used to correct for skewed data distributions. These two equations appear to be very similar, and a t-test of their slopes showed no significant differences between them, at the 5% level of significance. However, a t-test of their intercepts showed these to be highly significantly different ($t_{(2)(49)}$ = 34.00, $P \ll 0.001$). A similar equation between salinity (in g L^{-1}) and conductivity (at 25°C) has been described for Australian salt lakes by Williams (1986), while Buckney and Tyler (1973a) found an approximately linear relationship between conductivity and ionic concentration in Tasmanian waters up to 300 mg L^{-1}.

4.3.5 Major Ions Present

The concentrations of the seven major ions are given in Tables 4.1 (for the Bass Strait islands) and 4.2 (for western and south-west Tasmania). An estimate of ionic imbalance, where total anions are expressed as a percentage of total cations, is also given. These imbalances may be attributed to interference by gilvin in determinations of calcium, and in particular of sulphate, causing an underestimation of this anion by the $BaCl_2$ method used. Some changes may also have occurred in bicarbonate concentrations during transport. However, the imbalances are usually not of sufficient magnitude to affect interpretations of these data.

Sodium was the dominant cation in all cases, with the cationic order of dominance in almost all lagoons being $Na^+ > Mg^{2+} > Ca^{2+} > K^+$. Exceptions were No Duck Lagoon on Flinders Island, Attrills Lagoon on King Island, and Nye Bay and Paradise Lagoons from western Tasmania, all of which had elevated levels of calcium, their cationic orders being $Na^+ > Ca^{2+} > Mg^{2+} > K^+$. Additionally, Lake Strahan on 9 November, 1984 had the cationic order $Na^+ > Mg^{2+} > K^+ > Ca^{2+}$. Chloride was the dominant anion in all lagoons. Twenty-two samples from western and south-west Tasmania, and sixteen from the Bass Strait islands had the anionic dominance order $Cl^- > SO_4^{2-} > HCO_3^-$, while $Cl^- > HCO_3^- > SO_4^{2-}$ was the anionic order in all the remaining lagoons. Groundwater from near Surprise Bay, King Island, was dominated by calcium and bicarbonate, but sodium still comprised one third of the total cations, and relative proportions of chloride indicate a considerable marine influence still present in its chemical composition.

The ionic compositions of the Bass Strait island lagoons are shown in the ternary diagrams of Figure 4.5, while those of western and south-west Tasmania are shown in Figure 4.6.

4.3.6 Dissolved Iron and Silica

Iron was measured only in the samples from the Bass Strait islands (Table 4.1), and values were all low, as expected from oxygenated waters. The greatest amount, 0.90 mg L^{-1}, was found in Sticks Lagoon on Flinders Island. The values are mostly equal to or lower than those reported by Buckney and Tyler (1973a) for waters from King and Cape Barren Islands. The three highest values in this study were all from lagoons with low pH, but not all acidic lagoons had similar levels.

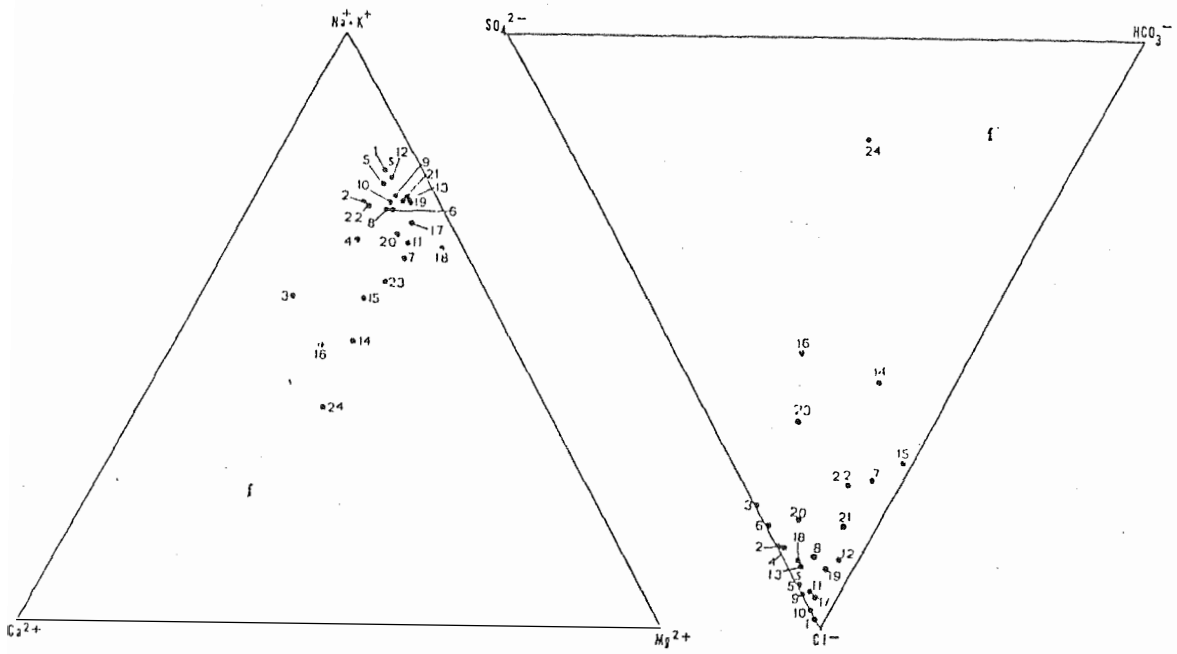


Figure 4.5 Ternary diagrams showing the ionic proportions of the lagoon waters from King and Flinders Islands, Bass Strait. Numbers refer to lagoons listed in Table 4.1. F = World Average Freshwater, s = seawater.

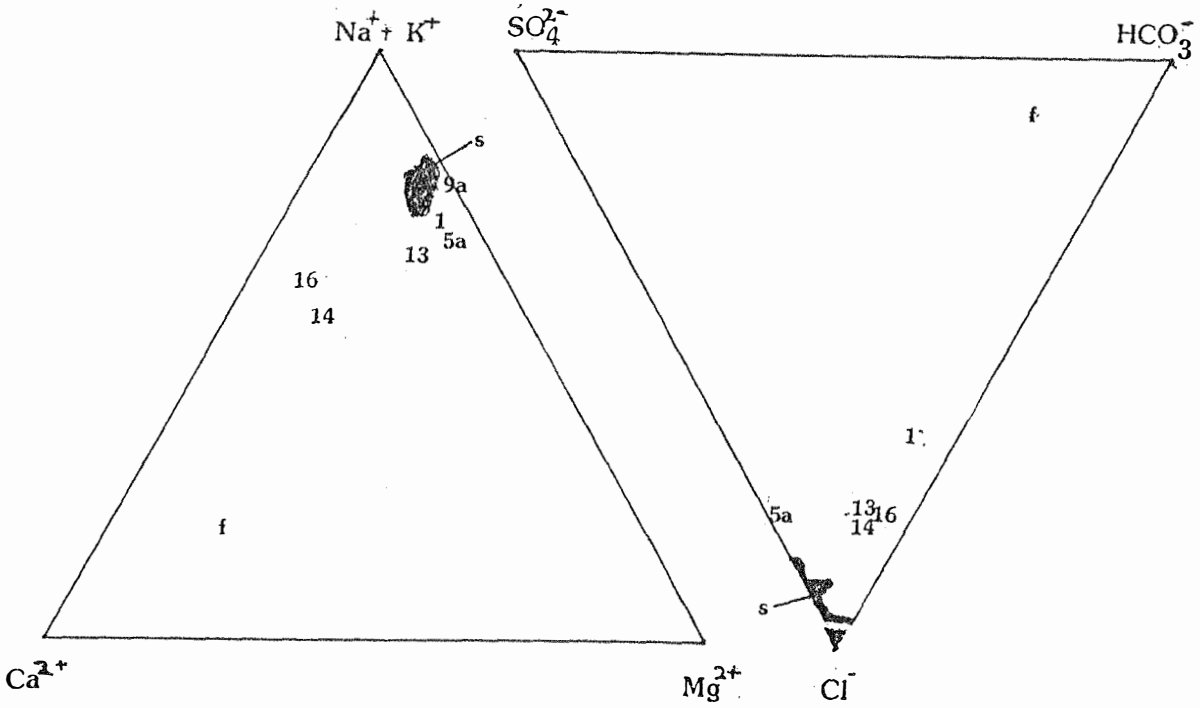


Figure 4.6 Ternary diagrams showing the ionic proportions of the lagoon waters from western and south-west Tasmania. The ionic proportions of most lagoons fall within the shaded areas. Others are indicated by number, as listed in Table 4.2. F = World Average Freshwater, s = seawater.

Silica concentrations of the Bass Strait island's lagoons (Table 4.1) were likewise low, with principal exceptions being Sticks and Sandy Lagoons on Flinders Island, while Ridge Lagoon had the highest levels of the King Island waters. Even silica levels in the groundwater were low. These data support those of Buckney and Tyler (1973a) for King Island lagoons, but give no indication of the extent of siliceous dunes in the lagoon's catchments.

Silica values from the western and south-west Tasmanian coastal lagoons were more variable (Table 4.2), ranging from less than 0.10 mg L^{-1} in Ocean Beach Lagoon, Teal Pond, and the dune ponds near New River Lagoon, to 4.18 mg L^{-1} in Lake Mallana. However, Gibbs Lagoon fell well outside this range, with a value of 19.8 mg L^{-1}. The median value for these waters was 0.62 mg L^{-1}.

4.3.7 Nutrient Analyses

Total phosphorus levels of the Bass Strait Island lagoons were mostly very high (Table 4.1), and they can be described as highly eutrophic (OECD, 1982). The highest levels occurred in the King Island lagoons, especially those from the southern end of the island, where considerable amounts of phosphorus were present, even in the groundwater. The variation amongst the lagoons possibly reflects different inputs of locally applied agricultural fertilizers in their catchments, or from the decay of surrounding vegetation, rather than direct geochemical influence. However even lagoons close together, such as the five at the north-eastern end of King Island, and which are in a nature reserve, show wide variation in total phosphorus levels. These local variations cannot be explained.

The total phosphorus levels of the lagoons from western and south-western Tasmania were much lower (Table 4.2), due to the undisturbed nature of the catchments of most. However several are still mesotrophic to eutrophic (O.E.C.D., 1982), with the highest value 58 $\mu g\ L^{-1}$, being measured in Lake Strahan.

Total nitrogen levels varied from 100 $\mu g\ L^{-1}$ in Freney Lagoon to 770 $\mu g\ L^{-1}$ in South Strahan Lagoon (Table 4.2), with a median of 325 $\mu g\ L^{-1}$. Based on Vollenweider's (1971) criteria, these total nitrogen data also indicate the lagoons to be mesotrophic to eutrophic.

4.3.8 Pearson Correlation Analysis

Pearson correlation analyses were undertaken on the physico-chemical parameters of the lagoons from the two study areas. Gibbs Lagoon, in south-west Tasmania, was considered very atypical compared to the remainder of lagoons from that area, and the results from it were therefore excluded from these analyses.

Several marked features are evident from these analyses. Firstly, strong positive correlations (significant at the 5% level) were found between conductivity and all seven major ions, in the waters of both study areas (Table 4.3), resulting from increased concentrations of solutes causing higher conductivities. Concentrations of all seven major ions increase in parallel, causing significant positive correlations between most of them.

The second major feature can be seen when the cations and anions are considered as relative proportions of total cations and anions, respectively. Positive correlations, significant at the 5% level or greater, usually occur between ions indicative of either marine (e.g. sodium and chloride) or geochemical (e.g. calcium, magnesium, bicarbonate, and sulphate) origins (Gibbs, 1970) (Table 4.3). Likewise, strong negative correlations exist between many of the marine and geochemical ions. These correlations show that geochemical enrichment took place in some of the lagoons, decreasing the relative proportions of ions of marine origin in their waters. They also show some differences occurred in this enrichment between the lagoons of the two study areas. Calcium and bicarbonate were the main ions responsible for the geochemical enrichment of lagoons from western and south-west Tasmania, but magnesium and sulphate also contributed significantly in those from the Bass Strait islands.

The pH values of the coastal lagoon waters of western and south-west Tasmania were also significantly positively correlated with conductivity, and with the concentrations of each of the major ions. In comparison, correlations between pH and these parameters (apart from bicarbonate) were not significant in the Bass Strait island lagoon waters. These differences probably result from the ionic concentrations of the lagoon waters of King and Flinders Islands being moderately high, and are therefore already buffered, so that further increases in salinity have no further effect on pH, whereas the more dilute waters of the western and south-west Tasmanian lagoons are less well buffered

Table 4.3: Pearson correlation coefficients (r) between various parameters measured from the coastal lagoon waters of King and Flinders Islands, and from western and south-west Tasmania. NS = Not significant. * Significant at the 5% level. ** Significant at the 1% level. *** Significant at the 0.1% level or greater.

PARAMETERS	BASS STRAIT IS.	S.W. TASMANIA	PARAMETER	BASS STRAIT IS.	S.W. TASMANIA	PARAMETERS	BASS STRAIT IS.	S.W. TASMANIA
			$Fe:\%Cl^-$	.409*	Not measured	$Fe:\%HCO_3^-$	-.404*	Not measured
$K_{18}:Na^+$	.989***	.976***	$Mg^{2+}:Cl^-$	.984***	.921***	$pH:K^+$	.290 NS	.670***
$K_{18}:K^+$	.998***	.910***	$Mg^{2+}:SO_4^{2-}$	.889***	.832***	$pH:Ca^{2+}$	.288 NS	.769***
$K_{18}:Ca^{2+}$	.890***	.486**	$Mg^{2+}:HCO_3^-$	.461*	.908***	$pH:Mg^{2+}$	.313 NS	.662***
$K_{18}:Mg^{2+}$	.984***	.947***	$Cl^-:SO_4^{2-}$	.883***	.914***	$pH:Cl^-$	.291 NS	.589**
$K_{18}:Cl^-$	.997***	.978***	$Cl^-:HCO_3^-$	.420*	.693***	$pH:SO_4^{2-}$	.257 NS	.652***
$K_{18}:SO_4^{2-}$	.910***	.914***	$SO_4^{2-}:HCO_3^-$	.370 NS	.777***	$pH:HCO_3^-$	.811***	.700***
$K_{18}:HCO_3^-$	.433*	.776***	$\%Na^+:\%Ca^{2+}$	-.887***	-.844***	$pH:\%Na^+$	-.370 NS	-.597**
$Na^+:K^+$	.990***	.899***	$\%Na^+:\%Mg^{2+}$	-.431*	-.008 NS	$pH:\%K^+$	-.410*	-.273 NS
$Na^+:Ca^{2+}$	.833***	.498**	$\%Na^+:\%Cl^-$	.883***	.551**	$pH:\%Ca^{2+}$	.378 NS	.578**
$Na^+:Mg^{2+}$	.984***	.976***	$\%Na^+:\%SO_4^{2-}$	-.726***	-.196 NS	$pH:\%Cl^-$	-.520**	-.764***
$Na^+:Cl^-$	.997***	.963***	$\%Na^+:\%HCO_3^-$	-.791***	-.567**	$pH:\%SO_4^{2-}$	.469*	.134 NS
$Na^+:SO_4^{2-}$	.864***	.927***	$\%Ca^{2+}:\%Mg^{2+}$	-.022 NS	-.507**	$pH:\%HCO_3^-$	.437*	.891***
$Na^+:HCO_3^-$	.424*	.845***	$\%Ca^{2+}:\%Cl^-$	-.819***	-.363 NS	$K_{18}:\%K^+$	-.214 NS	-.399*
$K^+:Ca^{2+}$	.876***	.479*	$\%Ca^{2+}:\%SO_4^{2-}$	.776***	-.059 NS	$K_{18}:\%Cl^-$	.173 NS	-.522**
$K^+:Mg^{2+}$	.985***	.870***	$\%Ca^{2+}:\%HCO_3^-$	.663***	.517**	$K_{18}:\%HCO_3^-$	-.360 NS	.607***
$K^+:Cl^-$	.996***	.915***	$\%Mg^{2+}:\%SO_4^{2-}$	.091 NS	.420*	$pH:g_{440}$	-.708***	-.481*
$K^+:SO_4^{2-}$	.917***	.877***	$\%Mg^{2+}:\%HCO_3^-$	.435*	.034 NS	$g_{440}:Ca^{2+}$	-.412*	-.176 NS
$K^+:HCO_3^-$	.424*	.692***	$\%Cl^-:\%SO_4^{2-}$	-.799***	-.641***	$g_{440}:HCO_3^-$	-.569**	-.250 NS
$Ca^{2+}:Mg^{2+}$	.808***	.411*	$\%Cl^-:\%HCO_3^-$	-.911***	-.812***	pH:Fe	-.540**	Not measured
$Ca^{2+}:Cl^-$	.866***	.437*	$\%SO_4^{2-}:\%HCO_3^-$	.481*	.073 NS	$SiO_2:SO_4^{2-}$	.315 NS	-.461*
$Ca^{2+}:SO_4^{2-}$	.892***	.420*	$pH:K_{18}$	.311 NS	.638***	g_{440}:Total N	Not measured	.811***
$Ca^{2+}:HCO_3^-$	.311 NS	.463*	$pH:Na^+$	.282 NS	.676***	Tn:Total P	.520**	Not measured

and therefore pH may rise as concentrations of solutes, in particular alkaline earth bicarbonates, increase.

Other significant correlations of note listed in Table 4.3 include those between total phosphorus and turbidity, g_{440} and total nitrogen, pH and iron, and pH and g_{440}. Phosphorus may be absorbed onto the surface of tripton particles, and in some cases turbidity may result from organic matter in the water, so that phosphorus and turbidity both increase together. A similar significant positive correlation was found between total phosphate and turbidity in New South Wales inland freshwaters (Chapter Three), as was one between g_{440} and total nitrogen. This was considered due to nitrogen comprising up to 5% of the elemental composition of dissolved humic substances (Schnitzer, 1978). The significant negative correlation between pH and iron may be due to iron being more soluble in acidic waters.

Significant negative correlations were found between water colour (= g_{440}) and pH in the waters of both study areas, as acidic dissolved organic substances depress pH. Similar negative correlations were found for the dune lakes of south-east Queensland (Chapter Three), and have been documented for coastal dune lakes elsewhere (Bayly, 1964; Bayly *et al*, 1975; Timms, 1982). Regression analysis showed the following relationships between pH and g_{440} in the three areas of this study:-

Equation 1). Western and south-west Tasmanian coastal lagoons.

$g_{440} = 45.5 - 5.65\ pH \quad (r^2 = 0.232, n = 24)$

Equation 2). King and Flinders Island coastal lagoons.

$g_{440} = 72.4 - 8.66\ pH \quad (r^2 = 0.501, n = 24)$

Equation 3). South-east Queensland freshwater coastal dune lakes.

$g_{440} = 23.4 - 3.64\ pH \quad (r^2 = 0.274, n = 26)$

t-tests shows no significant differences between the slopes of Equation 1) and 2); or between Equation 1) and 3), but the slopes of the regression equations for the Bass Strait island lagoons (Equation 2) and the south-east Queensland lakes (Equation 3) were significantly different ($t_{(2)(46)} = 2.07$; $0.05 \lessapprox P \lessapprox 0.02$). However, additional t-tests of the intercepts of Equations 1) and 2), ($t_{(2)(45)} = 3.10$; $0.01 \lessapprox P \lessapprox 0.001$), and Equation 1) and 3) ($t_{(2)(47)} = 8.96$; $P \lessapprox 0.001$) were highly significant. Thus, although significantly correlated, the responses of pH to changes in g_{440} are sufficiently different in all three areas, due probably to differences in ionic concentrations causing different buffering effects.

4.3.9 Principal Co-ordinates Analyses of the Lagoons from the Bass Strait Islands

Many King Island lagoons are in close proximity to each other, but despite this show considerable differences in their physiochemical properties. A principal co-ordinates analysis (Gower, 1966) was used to display this variability, and also differences between lagoons from both King and Flinders Islands.

An ordination showing the distribution of lagoons in the space defined by the first two coordinate axes is shown in Figure 4.7, and the principal component loadings for the parameters used are given in Table 4.4. Although most parameters contributed, the first coordinate axis reflects increasing geochemical influences and changing pH, iron concentration, turbidity, and water colouration, while the second coordinate axis mainly describes variability due to conductivity, silica, total phosphorus, and the two optical parameters.

Although the ordination revealed a continuous variation spread over both axes, some groupings of the lagoons are apparent. It effectively separates the Flinders Island from the King Island lagoons, on the second coordinate axis, although Sticks Lagoon (No. 1) is also separated from the other five Flinders Island lagoons, due to its low pH and conductivity, and higher iron, total phosphorous, percent sodium, and percent chloride. For the King Island samples, the highly coloured or turbid lagoons with high levels of total phosphorus form one distinct grouping, while the less coloured and turbid waters with some geochemical enrichment formed a second group. A third group, also with lower colour and turbidity, but with higher proportions of ions of marine origin, lies between them. Pearshape Lagoon (No. 15) was separated from all groups due to its high total phosphorous levels, and the groundwater sample (No. 24) was also divided from the King Island lagoons by the dominance of alkaline earth bicarbonates in it. However, this ordination reveals only 56% of the total variability amongst the samples (Table 4.4).

4.3.10 Phytoplankton Present in the Coastal Lagoons of Western and South-west Tasmania

Net phytoplankton were collected from eighteen of these lagoons during January and February, 1987 (Table 4.5). Most of the algae have been identified only to the genus level.

Botryococcus braunii Kützing was the most widely distributed taxon,

Table 4.4: Principal component loadings for coordinate axes one and two

Parameter	First coordinate axis	Second coordinate axis
Turbidity	0.16	0.25
g_{440}	0.22	0.35
pH	-0.31	-0.24
K_{18}	0.02	-0.50
Fe	0.23	-0.03
SiO_2	0.10	-0.26
Total P	0.10	0.36
$\%Na^+$	0.38	-0.20
$\% K^+$	0.17	0.20
$\% Ca^{2+}$	-0.36	0.00
$\% Mg^{2+}$	-0.14	0.38
$\% Cl^-$	0.41	-0.11
$\% SO_4^{2-}$	-0.35	-0.11
$\% HCO_3^-$	-0.36	0.24
Cumulative % of variation explained	37.52	55.82

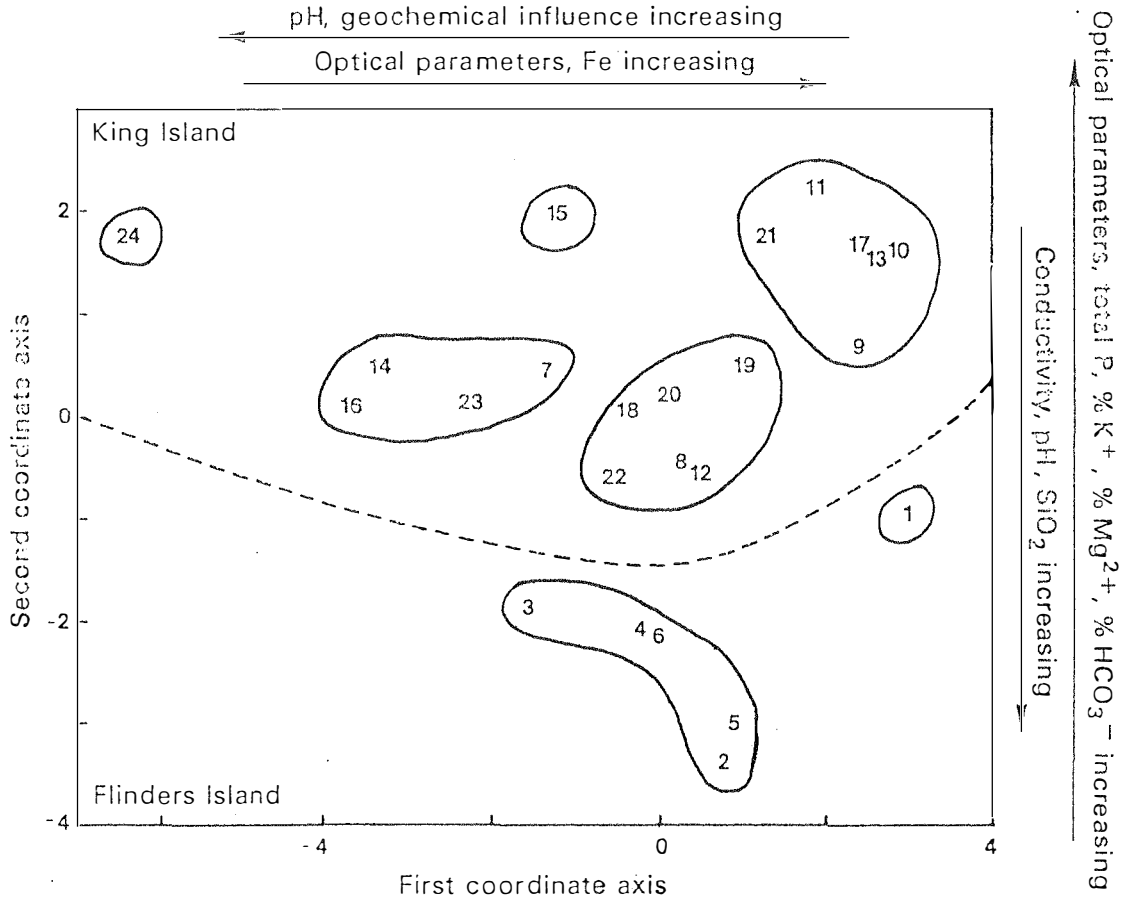


Figure 4.7 Ordination, by principal coordinates analysis, showing positions of the lagoons from King and Flinders Islands, Bass Strait, in the space defined by the first two coordinate axes. Numbers refer to the lagoons listed in Table 4.1. Dotted line emphasizes the separation of the two islands, while the solid lines emphasize divisions within the islands.

occurring in fourteen of the lagoons. Desmids were also very common and widespread, especially a biradiate species of *Staurastrum*, which is probably an undescribed species (P.A. Tyler, pers. comm.). Several other species of *Staurastrum* were also present, while *Closterium* spp., *Cosmarium* spp., *Staurodesmus* spp., and *Xanthidium* sp. were other commonly occurring desmids. Other green algae which occurred frequently included a colonial, coccoid green alga of the *Sphaerocystis* type, and *Asterococcus superbus* (Cienk) Scherf.

Taxa from other phycological divisions were also common, in particular various species of *Mallomonas* (Chrysophyceae), which were present in eleven of the lakes. The three most frequently encountered species were *M. splendens* (G.S. West) Playfair, *M. tasmanica* (Croome & Tyler) Asmund & Kristiansen, and *M. morrisonensis* Croome & Tyler. Species of *Peridinium* (Dinophyceae) were present in eleven lagoons, while various pennate and centric diatoms (Bacillariophyceae) occurred in fifteen. These included *Melosira granulata* (Ehrenberg) Ralfs., *Pinularia* sp., *Cymbella* sp., *Navicula* spp., *Frustulia rhomboides* (Ehrenberg) Di Toni, while *Asterionella formosa* Hass. was present in Gibbs Lagoon. Euglenoids were also present in some of the lagoons, but cyanobacteria were rarely encountered.

A number of species of freshwater flagellates appear to be confined almost entirely to these coastal lake environments (Croome and Tyler, 1987a; Croome *et al.*, 1987; Croome *et al.*, in press; Ling *et al.*, in press). These include the dinoflagellates *Thecadiniopsis tasmanica* Croome, Hallegraeff, and Tyler, present in eight of the lagoons (see Table 4.5), and *Prorocentrum playfairi* Croome and Tyler, found in six of the study sites; and a newly described chrysophyte, *Dynobryon unguentariforme* Croome, Ling, and Tyler, which occurred in Freney and Miller Lagoons.

The greatest range of phytoplankton occurred in Mulcahy Bay Lagoon, while Lake Bantic and Freney, Miller, and Big Lagoons also had considerable numbers of different taxa. In comparison, Lake Mallana, Gibbs Lagoon, and Heron and Teal Ponds were depauperate in phytoplankton, and a virtually monospecific bloom of *Mallomonas tasmanica* was present in Lake Koonya. However, some species, especially nanoplankton and picoplankton, may have been missed by the sampling method used. The contribution of these algae to community diversity and abundance is unknown.

Table 4.5: Phytoplankton present in the coastal lagoons of western and south-west Tasmania. Presence only is indicated.

Lake No.	LAKE / ALGA	*Eudorina elegans* Ehrenberg	*Pandorina morum* (Müller) Bory	*Asterococcus superbus* (Clenk.) Scherf.	cf. *Sphaerocystis*	*Ankistrodesmus falcatus* (Corda) Ralfs.	*Oocystis* sp.	*Botryococcus braunii* Kützing	*Dictyosphaerium* sp.	*Actinastrum hantzschii* Lagerheim	*Coelastrum* sp.	*Pediastrum duplex* Meyen	Unknown Filamentous green alga	Unidentified Saccoderm desmid	*Spirogyra* sp.	*Closterium* spp.	*Cosmarium* spp.	*Euastrum* spp.	*Microsterias* spp.	*Pleurotaenium* sp.	Filamentous desmid (*Sphaerozosma*?)	*Staurastrum* spp.	*Staurodesmus* spp.	*Triploceros gracile* Bailey	*Xanthidium* sp.	*Euglena* sp.	*Trachlemonas volvacina* Ehrenberg	*Prorocentrum playfairi* Croome & Tyler	*Peridinium* spp.	*Thecadiniopsis trasmanica* Croome, Hallegraeff & Tyler	*Dynobryon* spp.	*Mallomonas* spp.	*Synura* sp.	*Melosira granulata* (Ehrenberg) Ralfs.	Centric diatoms (unidentified)	*Asterionella formosa* Hass.	*Frustulia rhomboides* (Ehrenberg) Di Toni	Pennate diatoms (unidentified)	*Merismopedia* sp.	*Oscillatoria* sp.	*Anabaena* sp.
2	Lake Mallana	×																																						
3	Lake Bantic		×						×																															
4	Lake Garcia			×	×			×								×	×																							
5	Lake Ashwood					×							×	×																										
6	Little Bellinger Lake				×			×															×	×	×	×	×													
7	Lake Koonya							×																			×	×	×	×	×										
8	Ocean Beach Lagoon				×		×	×								×						×	×	×																
9	Lake Strahan			×				×															×	×	×															
10	South Strahan Lagoon							×							×								×	×																
11	Heron Pond				×			×																																
12	Teal Pond							×											×																					
14	Nye Bay Lagoon						×	×				×					×	×																						
15	Mulcahy Bay Lagoon			×			×	×										×			×																		
16	Paradise Lagoon							×			×	×						×	×																					
17	Freney Lagoon			×				×										×	×																					
18	Millar Lagoon							×		×			×							×																				
24	Gibbs Lagoon																										×												
25	Big Lagoon	×	×	×	×	×	×	×																																

Chlorophyceae | Euglenophyceae | Dinophyceae | Chrysophyceae | Bacillariophyceae | Cyanophyceae

4.4 DISCUSSION

4.4.1 The Physiochemical Properties of the Coastal Lagoons

The coastal lakes and lagoons of the Bass Strait islands and western and south-west Tasmania are typically non-turbid and dystrophic. Although turbid waters are commonplace on the Australian mainland (Kirk, 1985), they are unusual in Tasmania (Chapter Three), and suspended particulate matter would be of importance limnologically in only three King Island lagoons. In contrast, although highly coloured coastal dune lakes have been reported from elsewhere in Australia, (Bayly, 1964; Timms, 1973, 1982, 1986b), the results of this study (and of Brand, 1967; and Buckney and Tyler, 1973a) indicate that some lagoons from the two study areas rank amongst the most highly coloured natural waters yet recorded. Their gilvin concentrations are considerably greater than those of coastal dune lakes from south-east Queensland and north-east New South Wales (Chapter Three), and also exceed those of the polyhumic forest lakes common to Finland (Arvola, 1983; Jones and Arvola, 1984; Salonen, 1984).

The high humic content of many of these lagoon waters causes the rapid attenuation of solar radiation. Although it was not possible to measure their underwater light climates, these can be estimated using the procedures developed in Chapter Three. With turbidity so low, measurements of gilvin alone may be used to calculate the expected mean downwelling vertical attenuation coefficients, $Kd_{(av)}$, for these waters. The least coloured of the lagoons, Lake Bantic, would have a $Kd_{(av)}$ value of approximately 0.33 m^{-1}, and an underwater spectral distribution of the light similar to that of Great Lake (see Chapter Three), with the wavelengths of maximum transmission centred around 570 to 580 nm. Mesohumic Rebecca and Penny Lagoons would have $Kd_{(av)}$ values of about 1.50 m^{-1}, while the higher g_{440} of Hibbs and Granite Lagoons would give them $Kd_{(av)}$ values of about 2.40 m^{-1}, and underwater spectral distributions like that of Lake Pedder (nova) (see Chapter Three), where only predominantly red light,

centred around 680 nm, remains. The polyhumic waters of this study would have underwater spectral distributions akin to that of Lake Chisholm, with light of 700 to 710 nm being transmitted best. Examples of $Kd_{(av)}$ values for some would be 4.04 m^{-1} for Dembys Lagoon, 4.67 m^{-1} for Teal Pond, 5.60 m^{-1} for Heron Pond, and 5.95 m^{-1} for South Strahan Lagoon. Should the relationship between ln $Kd_{(av)}$ and ln g440 (Equation 4, Table 3.3 of Chapter Three) remain linear up to the g440 value of 253.33 m^{-1} recorded for Gibbs Lagoon, this would give it a $Kd_{(av)}$ of 13.42 m^{-1}. The attenuation of light in the majority of these coastal lakes and lagoons is considerably greater than that measured in similar lakes from south-east Queensland (Chapter Three). However, the attenuation calculated for extremely dystrophic Gibbs Lagoon is still much less than that measured in extremely turbid Yarrie Lake in New South Wales (Chapter Three), suggesting that high levels of turbidity may have a greater effect on light attenuation than high concentrations of humic substances.

Summer thermal measurements made on a few lagoons from western Tasmania showed only one, Lake Garcia, developed definite thermal stratification. Although the remainder displayed steep thermal gradients, high levels of oxygen in their bottom waters show this would be transitory, and polymixis would be normal, due to their shallowness and exposure to considerable amounts of wind. Coastal lakes from northern New South Wales (Timms, 1969), Fraser Island, Queensland (Miller et al, 1976; Bayly et al, 1975; Arthington et al, 1986), and from Georgia, U.S.A. (Stoneburner and Smock, 1979) have also been shown to be predominantly polymictic.

The steep thermal gradients would result from the high humic contents of these lagoons. Light is absorbed by gilvin and converted to heat energy close to the surface, warming these waters considerably, while the bottom waters remain cold. This occurs especially on hot, sunny days, but the stratification would probably diminish as the lagoons cool at night, and they would circulate completely during intervening periods of dull, windy, and possibly wet weather. Such thermal patterns would be similar to the daily regime of stratification and circulation shown by Northern Territory billabongs (Walker, 1984). In comparison to the highly dystrophic lakes, Lake Bantic showed no signs of a thermal gradient, as its transparent waters allow light to penetrate right to its bottom, thus creating an even distribution of heat energy throughout its entire depth.

However, thermal stratification may occur if lakes are sufficiently well sheltered from wind action, and this would probably account for the warm monomictic nature of Lake Garcia, and the almost complete depletion of oxygen in the bottom waters of Little Bellinger Lake. Both are surrounded by forests and hills. Similar stratification occurs in Brown Lake, North Stradbroke Island (Bensink and Burton, 1975), and Hidden Lake, Fraser Island (Longmore et al, 1983; Torgerson and Longmore, 1984). The importance of shelter allowing stable thermal stratification is further illustrated by the studies reported in Chapter Five.

Dissolved humic substances may also contribute considerably to the acidification of natural waters (Glover and Webb, 1979; Henricksen and Seip, 1980). This acidification was particularily so in the lagoon waters of south-west and western Tasmania, where salinities were low enough to prevent buffering. pH values here were typical of those from the dilute coastal lakes of north-east New South Wales and of Queensland (Chapter Three; and Bayly, 1964; Bayly et al, 1975; Bayly and Williams, 1972; Bensink and Burton, 1975; Timms, 1969, 1982, 1986b; Bensink, 1976; Arthington and Watson, 1982). However, those of King and Flinders Islands were higher, and more similar to the pH's reported for some of the more saline Australian coastal lakes (Bayly and Williams, 1966; Timms, 1973, 1977b; Williams and Buckney, 1976; Congdon and McComb, 1975; Gordon et al, 1981).

The water chemistry of the lagoons from western and south-west Tasmania follows the same pattern found (Buckney and Tyler, 1973a,b) for most other inland waters of the area. Although dilute, they are generally less so than many coastal lakes from eastern Australia, such as those from Fraser Island (Bayly, 1964 - see also Chapter Three), or northern New South Wales (Timms, 1969, 1982, 1986a). In comparison, the lagoons from the Bass Strait islands had considerably higher salinities. These too were more saline than nearby coastal lagoons in eastern and western Victoria (Timms, 1973, 1977b), but had salinities similar to some from the eastern and north-east coasts of Tasmania (Buckney and Tyler, 1973a, 1976; Croome and Tyler, 1987a) and south-west Western Australia (Gordon et al, 1981; Newman and Hart, 1984; Congdon and McComb, 1976). These higher salinities may be explained by the high westerly wind regime experienced by the west and south-west coasts of Tasmania, and the Bass Strait islands depositing greater amounts of cyclic salts within their lagoon

catchments. This would be coupled with annual evaporation exceeding precipitation on the Bass Strait islands (Buckney and Tyler, 1976), which would increase the salinities of their lagoons still further. However, even in the most saline lagoons of Flinders Island, evaporative concentration is still insufficient to produce the salinities typical of many coastal lakes of South Australia (Bayly, 1970; Bayly and Williams, 1966; Williams and Buckney, 1976), or of Rottnest Island (Edward, 1983, Bunn and Edward, 1984).

The $Na^+ > Mg^{2+} > Ca^{2+} > K^+$ pattern of cationic dominance typical of the lagoons from the two areas of this study, is also common in many Australian coastal lakes (Bayly, 1964; Bayly and Williams, 1966, 1972; Timms, 1969, 1973, 1982, 1986b; Williams and Buckney, 1976; Congdon and McComb, 1976), exceptions being in some from western Victoria (Timms, 1977) and Cape York (Timms, 1986b), and in some calcium dominated lagoons in northern New South Wales (Timms, 1982). Buckney and Tyler (1973a, 1976) reported several lagoons from King and Cape Barren islands where $Na^+ > Ca^{2+} > Mg^{2+} > K^+$, and one, Pearshape Lagoon, where calcium was the dominant cation. This contrasts with the results from this lagoon from this study, where calcium was third in order of cations. The differences may be temporal, depending on sampling time, or resulting from the dry summer prior to sampling for this study. The chemistry of the lagoons near Strahan was similar to that reported previously (King and Civil Investigation Division, Hydro-Electric Commission, 1978b) for three of them.

The $Cl^- > SO_4^{2-} > HCO_3^-$ order of anionic dominance is also predominant in coastal lake waters elsewhere in Australia, although the $Cl^- > HCO_3^- > SO_4^{2-}$ order occurs frequently too (Bayly, 1964; Bayly and Williams, 1966, 1972; Timms 1973, 1977, 1982, 1986b; Williams and Buckney, 1976), and both anionic orders have been reported previously from lagoon waters from the Bass Strait islands and other coastal areas of Tasmania (Buckney and Tyler, 1973a, 1976; King and Civil Investigation Division, Hydro-Electric Commission, 1977b). However, bicarbonate dominated coastal lake waters occur in western Victoria, east Gippsland, and northern New South Wales (Timms, 1973, 1977b, 1982).

Although the chemistry of the lagoons was dominated by ions of oceanic origin (Gibbs, 1970), most showed at least slight enrichment with calcium, magnesium, and bicarbonate, in particular No Duck Lagoon on Flinders Island; Woodlands, Pioneer, Attrills and Sullivans

Lagoons and Lake Flanigan on King Island, and Rebecca, Hibbs, Nye Bay, and Paradise Lagoons from western Tasmania. The groundwater sample from King Island showed the closest affinity to a World Average Freshwater type ionic distribution. Such enrichment was due to localized variations resulting from the nature of the surrounding dune material. Brand (1967) found elevated levels of calcium in King Island lakes surrounded by calcareous dunes.

Silica values were mostly too low to reveal any connection between siliceous dunes and water chemistry. Concentrations of the south-west and western Tasmanian lagoons were mostly higher than those of the Bass Strait islands, and also exceeded those of the Fraser Island lakes (Little and Roberts, 1983), although silica concentrations of lake waters from the Cooloola Sand Mass were greater (Reeve *et al.*, 1985). Low silica concentrations are considered indicative of short residence times for the waters of coastal dune lakes, while high silica contents indicate equilibrium with the surrounding siliceous dunes (Little and Roberts, 1983).

Ordination procedures appear to be rarely used in physiochemical limnology. Estrada (1975) successfully characterized differences between Spanish reservoirs using principal components analysis, while Ilmavirta *et al.*, (1984) were able to divide Finnish lakes into eight groups based on their physiochemical properties, using the same method. Differences between lake waters and those of various creek catchment basins on Fraser Island have also been highlighted (Little and Roberts, 1983) using canonical variates and discriminant analysis. In this study, a principal coordinates analysis revealed limnological differences betwen waters of the Bass Strait Islands. Whether this is matched by a biotic dissimilarity is unknown.

4.4.2 The Phytoplankton of the Lakes of Western and South-west Tasmania

Although only a brief examination of the phytoplankton of these coastal lagoons, the results indicate that desmids constitute the majority of the green algal taxa present, and apart from the diatoms and the few cyanobacteria, all the non-chlorophyte algae present were flagellates. However, the results of this study are those for single samples only, obtained about mid-summer, and different algal taxa may well be present at other times of the year. No additional data are available to determine if the composition of the phytoplankton communities of these waters changes seasonally.

Flagellates are considered to be at an ecological advantage in humic acid waters (Ilmavirta, 1984), and those of the Divisions Chrysophyceae and Cryptophyceae have been found to dominate the phytoplankton communities of Finnish polyhumic lakes (Ilmavirta, 1984; Ilmavirta *et al.*, 1984). Many dystrophic Tasmanian lakes, especially from coastal areas, also contain rich collections of phytoflagellates, especially chromophytes (Croome and Tyler, 1987b). Likewise, dilute acidic waters with low calcium and magnesium concentrations are also considered a typical habitat for desmids (Lee, 1980). However, the data from this survey are insufficient to conclude whether these ecological factors have any influence on the phytoplankton present in these western and south-west Tasmanian coastal lakes.

Freshwater acidic coastal dune lakes along the entire east Australian seaboard have a homogeneity in their fauna, with many species virtually restricted to such habitats (Arthington, 1977; Timms, 1986b). Recently new species of phytoflagellates, including some of great rarity, have been described from humic, acidic Tasmanian coastal lagoons which occur predominantly within this type of habitat (Croome and Tyler, 1987a, b; Croome *et al.*, 1987; Croome *et al.*, in press). Some were observed in this study (Section 4.3.10). One, the dinoflagellate *Prorocentrum playfairi*, was reported from the Sydney area (as *Exuviaella lima* (Ehr.) Schutt.) by Playfair (1919), and has been found in freshwater coastal dune lakes in north-east New South Wales (Chapter Three). Another, the chrysophyte *Dynobryon unguentariforme*, has been found in some Moreton and Fraser Island lakes and swamps (Croome *et al.*, in press). The wide distribution of these two phytoplankters, and of the copepod *Calomoecia tasmanica* (Smith), which is also present in most coastal lagoons, suggests that the freshwater coastal lakes and lagoons of western and south-west Tasmania display strong limnological affinities with those from other

parts of eastern Australia.

4.5 CONCLUSIONS

The study shows that the water chemistry of the lagoons from western and south-west Tasmania, and from the Bass Strait islands, is similar to that shown by the majority of freshwater coastal dune lakes of the eastern Australian seaboard (e.g. Bayly, 1964; Bensink and Burton, 1975; Timms, 1969, 1982, 1986a, b), although their salinities, and in particular those of King Island, are somewhat higher. Some Flinders Island lagoons are saline, and resemble coastal salt lakes from South Australia and Western Australia instead. While some localised enrichment with alkaline earth bicarbonates occur in those located in calcarious dunes, the maritime influences on the study areas are particularly strong, and ions of oceanic origin (Gibbs, 1970) dominate the water chemistry. Many of the lagoons were highly dystrophic, far more so than their mainland Australian counterparts (Bayly, 1964; Timms 1973, 1982, 1986b), and many were eutrophic, but few were turbid. However, the lagoons displayed a considerable variability in their physiochemical features, and even those in close proximity could be markedly different.

The high dystrophy of many of these waters serves to highlight the major effects of dissolved humic substances within them. Firstly, it causes the very rapid extinction of solar radiation penetrating the surface, resulting in turn in extremely shallow euphotic depths and spectral distributions centred on red light. Severe thermal gradients may also result from this, although these are usually of a transient nature, dependent on prevailing meteorological conditions. Secondly, the high concentrations of dissolved humic substances leads to a marked reduction of pH in many of these waters, in particular in the more dilute, and therefore unbuffered, of them.

Although subject to the limitations of the sampling methods employed, a phytoplankton flora dominated by desmids and flagellates is most probable in the majority of dystrophic freshwater lagoons from western and south-west Tasmania. Some phytoflagellate species are possibly virtually restricted to such habitats. Because some of these species also occur in similar coastal lagoons elsewhere in eastern Australia, it is hypothesized that all these waters may exhibit floristic affinities, similar to those attributed to their fauna by Arthington (1977) and Timms (1986a). These would also include the lagoons from King and Flinders Islands.

CHAPTER FIVE

DETAILED LIMNOLOGICAL STUDIES OF DYSTROPHIC LAKES AND RESERVOIRS FROM WESTERN TASMANIA

5.1 INTRODUCTION

5.1.1 The Aims and Scope of these Studies

This chapter deals with three separate, detailed studies of dystrophic lakes and reservoirs from the western limnological province of Tasmania (see Section 2.1.1), and demonstrates the effects dissolved humic substances have on the limnology of such waterbodies. Although individual studies, when considered together they also represent a progressive series illustrating many of the features required for the establishment and maintenance of meromixis.

Although Tasmania has many humic lakes (Chapter Two, Three and Four; and Buckney and Tyler, 1973a,b), few are as highly dystrophic as the small forest lakes of Finland (Section 2.4.6). Exceptions include the lakes along the lower Gordon River (see below), and some shallow coastal lagoons (Chapter Four). Lake Chisholm is the only non-meromictic, polyhumic forest lake in Tasmania, and thus offers an ideal site for physicochemical and biological studies of highly humic waters, comparable to the Finnish examples. Other limnological influences, such as basin morphometry and the shelter afforded by surrounding forest and hills, can also be assessed. These studies constitute the first part of this chapter.

Investigation of four humic reservoirs of the Pieman River Power Development form the second part of this chapter. Tyler (1980) predicted the onset of chemical stratification in impoundments occupying steepsided, heavily vegetated river valleys, including those of the Pieman River, and this study tests the accuracy of that prediction. Comparative data were also collected from Lake Barrington, the first meromictic Tasmanian reservoir (Tyler and Buckney, 1974). Dissolved humic substances may play an important role in the physicochemical limnology of these reservoirs, and this too was considered during this study.

Three of the four small, highly dystrophic lakes on the eastern bank of the lower Gordon River were found to be ectogenically meromictic by King (1980) and King and Tyler (1981a, 1982a, 1983). Changes in the flow regime of the Gordon River, caused by the commissioning of a large hydro-electric dam on its middle reaches (King and Tyler, 1982b) has lead to the decline of meromixis in these lakes. By 1986, only one, Lake Fidler, remained meromictic, and even this had changed considerably from its original condition (Tyler, 1986; Croome and Tyler, in press). The best manifestation of their meromictic condition was the extremely rapid change in apparent redox potential, from oxygenated to reducing conditions, over a depth of just centimetres, within the chemocline. This redoxcline has progressively deepened over a period of years, as meromixis has been erroded away, and therefore is a very good indicator of the demise of meromixis in the lakes. Additionally, the meromictic stabilities of the lakes also fall as the chemical stratification of the lakes decays.

The third section of this chapter employs these features to detail the demise of meromixis in these lakes. This section also allows refinement of the theory of the origins of their chemical stratification (King and Tyler, 1981a; Bowling, 1981; Tyler, 1986). However, the effects of other factors, such as lake morphology, shelter, and particularly the role of humics in the limnology of the lakes are also demonstrated. These factors may assist the maintenance of meromixis within the lakes, and therefore slow its demise. Thus, the lakes provide a counter-point for those of the chemically stratified reservoirs of the Pieman River, and polyhumic Lake Chisholm. The great significance of the Gordon River lakes to the World Heritage Area is also highlighted, and a knowledge of the factors affecting the dynamics of these humic, meromictic lakes is sought from this study, to enable their effective management.

5.1.2 The Study Areas

The locations of the three study areas are shown in Figure 5.1. Lake Chisholm is situated in the far north-west of Tasmania, close to the southern bank of the Arthur River, 90 metres above river level and 120 metres above sealevel. The lake is well sheltered by surrounding hills, which form its small catchment. There are no inflow creeks, but a single outflow drains the lake northwards to the river. The lake is a sinkhole, a depression caused by a collapse in a local area

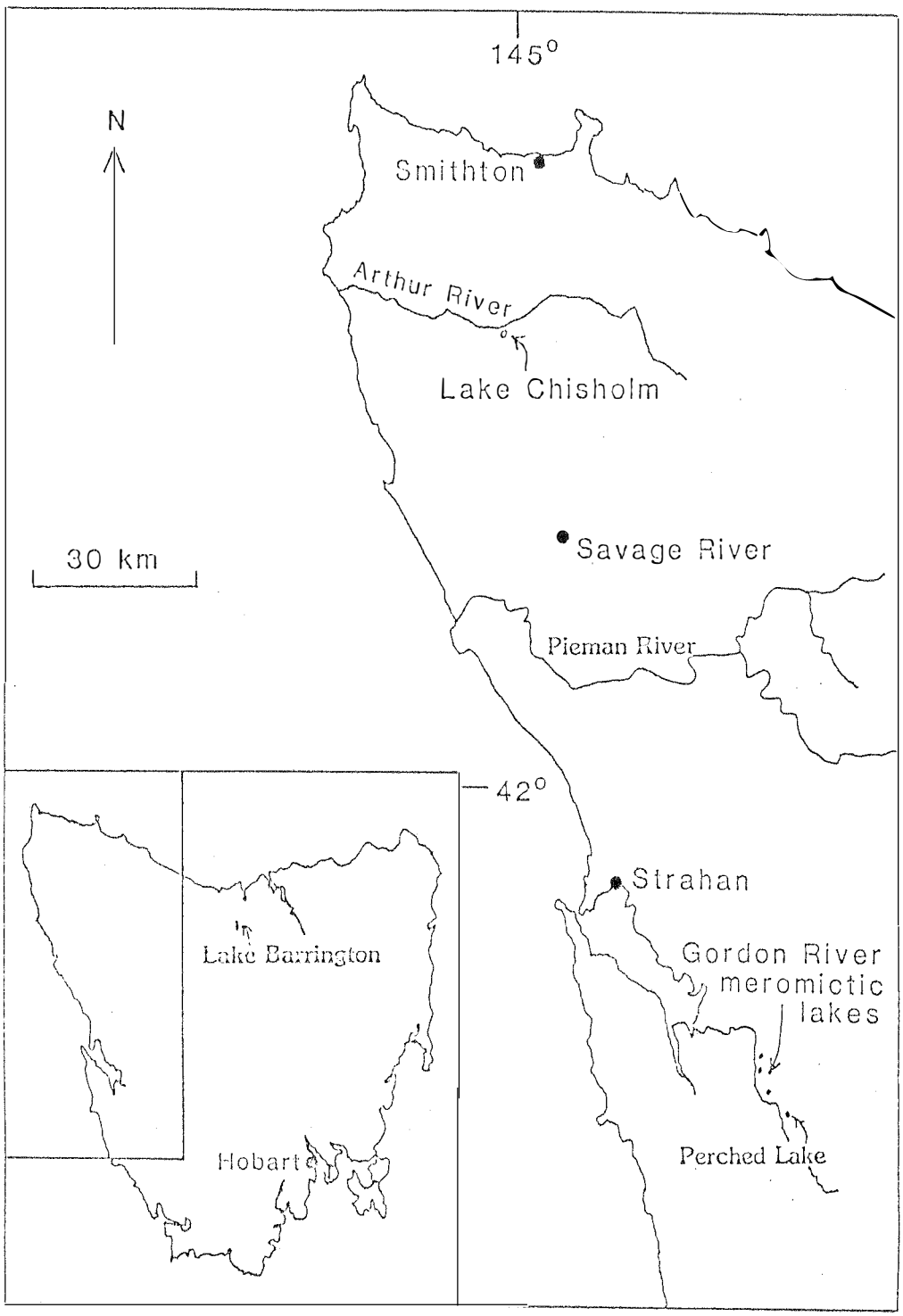


Figure 5.1 Map of Tasmania, showing the locations of the lakes and reservoirs of this study. See Figure 5.3 for more detail of the Pieman River Power Development.

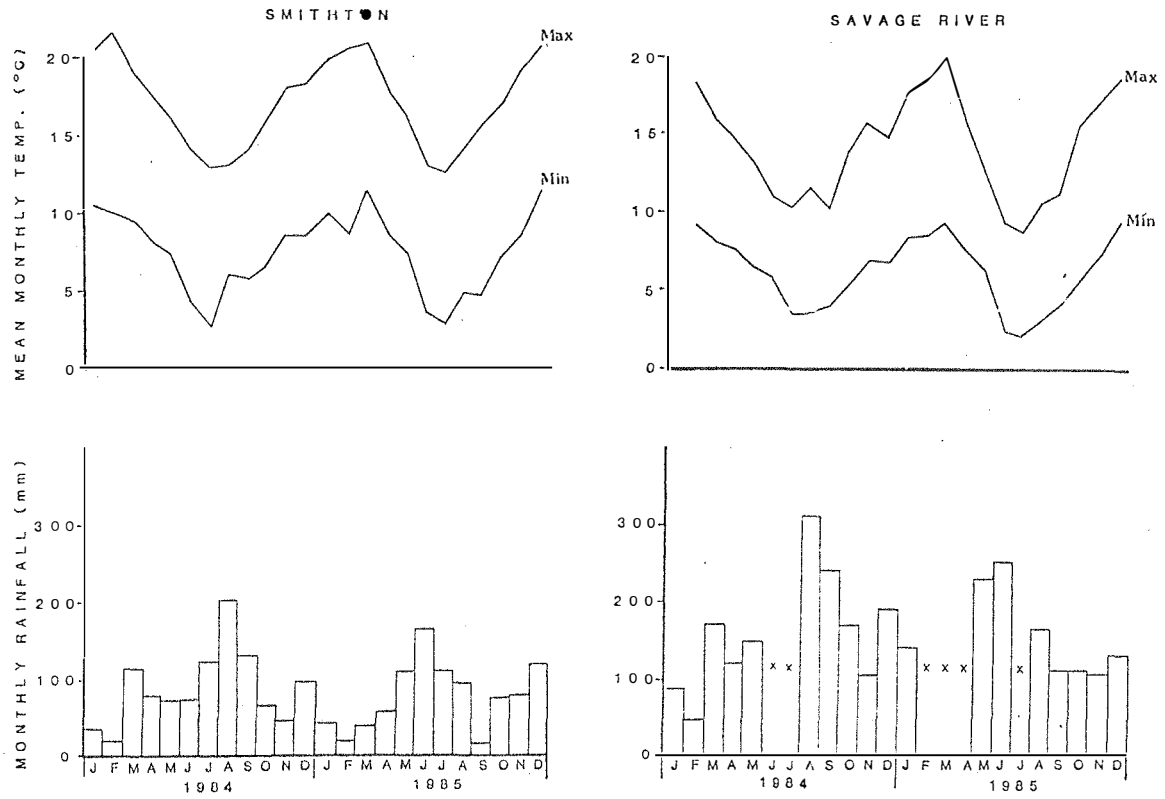


Figure 5.2 Mean monthly maximum and minimum air temperatures, and mean monthly rainfall for two stations in the vicinity of Lake Chisholm (see Figure 5.1). X = no data. Data courtesy of Commonwealth Bureau of Meteorology, Hobart.

of limestone. The area has a moist, maritime climate typical of all western Tasmania, with prevailing westerly winds. Available climatic data for the nearest stations (Smithton and Savage River - see Figure 5.1) are given in Figure 5.2.

Lake Chisholm is surrounded by thick "mixed" forest (*sensu* Gilbert, 1959), with the dominant trees being *Eucalyptus obliqua* L'Herit. and *E. viminalis* Labill., and a secondary tree layer of cool temperate rainforest species, in particular *Nothofagus cunninghamii* (Hook.) Oerst., but also with some *Atherosperma moschatum* Labill., *Eucryphia lucida* (Labill.) Baill., *Phyllocladus asplenii folius* (Labill.) Hook. F., and *Acacia melanoxylon* R.Br. The shrub layer contains *Anopterus glandulosus* Labill., *Cenarrhenes nitida* Labill., *Melaleuca squarrosa* Donn. ex. Sm., *Cyathodes juniperina* (Forst.) Druce; *Monotoca glauca* (Labill.) Druce, and the tree fern *Dicksonia antarctica* Labill. The ground cover is composed almost entirely of the fern *Blechnum wattsii* Tindale. This forest produces a deep peat which is responsible for the allochthonous organic material giving the lake its dark brown colouration. There is no emergent or aquatic vegetation around the waters edge.

The Pieman River is one of the main river systems of western Tasmania, being formed by the confluence of the Mackintosh and Murchison Rivers. Their source in the Cradle Mountain - Lake St. Clair National Park receives an annual rainfall in excess of 2000 mm, often as heavy winter snowfalls. Before impoundment the river valleys were narrow, steepsided, and heavily vegetated. Pre-impoundment water chemistry (Buckney and Tyler, 1973a; King and Civil Investigation Division, Hydro-Electric Commission, 1978a) showed their waters to be humic and of low salinity, with pH values between 6.0 and 7.5. Murchison River waters were dominated by sodium and chloride, but those of the Mackintosh and Pieman Rivers were considerably enriched with calcium and bicarbonate.

Of the four impoundments (Figure 5.3a), Lake Mackintosh, formed in August, 1980, occupies the former valleys of the Mackintosh and Sophia Rivers, while Lake Murchison was created in August, 1982, by daming the Murchison River. Lake Rosebery occupies the upper reaches of the Pieman River and the lower reaches of the Mackintosh and Murchison Rivers, and first filled in August, 1983, while Lake Pieman, further downstream, commenced spilling in May, 1986. All the

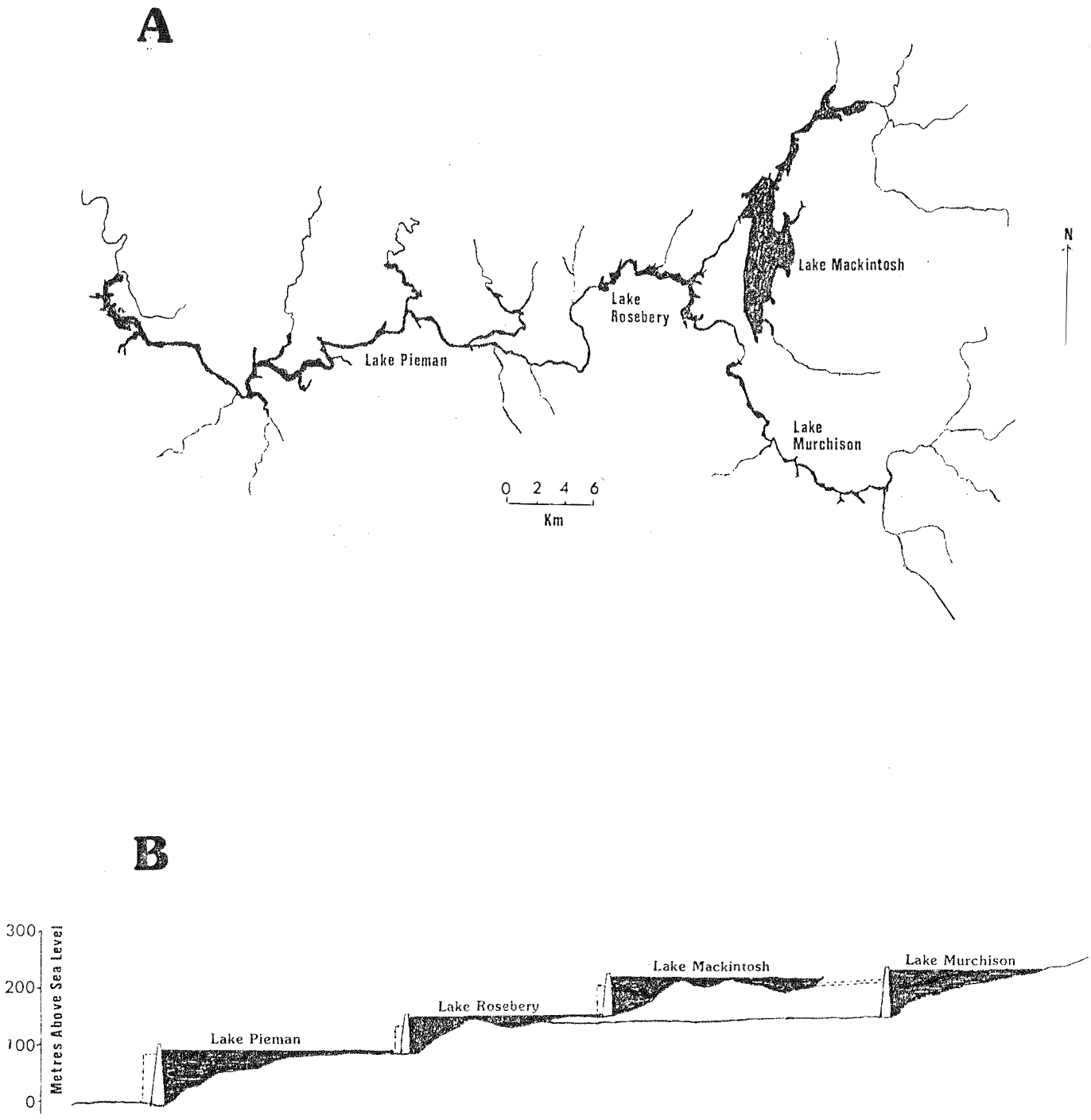


Figure 5.3 A. Map of the impoundments of the Pieman River Power Development. B. Vertical profile. Note the presence of high-level offtakes at each dam. Vertical scale 1:2000, horizontal not to scale (after Civil Engineering Branch, Hydro-Electric Commission of Tasmania, 1980).

reservoirs have high level offtakes, leaving a considerable amount of dead water between these and their bottoms (Figure 5.3b). Morphometric data is given by Peterson and Missen (1979).

Lake Barrington, on the Forth River in north-west Tasmania, is not part of the Pieman River Power Development, but also occupies a narrow steepsided river valley which was heavily vegetated prior to flooding. The reservoir receives water from Lake Cethana, another hydro-electric impoundment immediately upstream. Lake Barrington's chemical stratification has been described by Tyler and Buckney (1974), Tyler (1980), and Fast and Tyler (1981).

The polyhumic lakes of the Gordon River have been described by King and Tyler (1981a,b; 1982a, 1983). The three meromictic lakes are backswamp lakes, at river level, but separated from it by silt levee banks, while the fourth, Perched Lake, is of uncertain origin, seventeen metres above sealevel. All are surrounded by cool-temperate rainforest. Vegetational and climatic details of the area were given by King and Tyler (1981a).

5.2 METHODS

5.2.1 Data Collection and Analysis

Thermal profiles were measured *in situ* with calibrated thermistors. Water samples were obtained by Van Dorn water bottle for dissolved oxygen determinations by the azide modification of the Winkler method (A.P.H.A., 1971), or for dissolved sulphides by the method of Tyler and Buckney (1974). Aliquots of these samples were taken for immediate electrometric measurement of apparent redox potential (Eh), pH, and electrical conductivity (as K_{18}). Additional samples were obtained from across the chemoclines of the Gordon River meromictic lakes for similar chemical analyses using close interval samplers (points five centimetres apart) described by Baker *et al* (1985b) and Croome and Tyler (1984a). Joint use has been made with Croome (1984) of data sets for 1982 and 1983 from Lake Fidler and Sulphide Pool.

Additional water samples were taken from Lake Chisholm and the Pieman River reservoirs for ionic analysis using the proceedures described in Chapter Four. Samples for iron and manganese were stored in ground-glass stoppered glass bottles, with all air bubbles excluded, until analysis by atomic absorption spectrophometry. Two drops of concentrated nitric acid were added to prevent the iron and

manganese precipitating. Turbidity, gilvin, and other optical parameters were measured by the proceedures detailed in Chapter Three, as were the phytoplankton.

Samples from Lake Chisholm for PO_4-P and NO_3-N (including any NO_2) where filtered immediately after sampling through acid cellulose filters and preserved with $HgCl_2$ on ice. PO_4-P was measured colourmetrically within two hours of sampling by the ascorbic acid method (A.P.H.A., 1971), and NO_3-N within twelve hours with sulphanilamide and N-(1-naphtyl) ethylenediamine dihydrochloride after colour removal with aluminium hydroxide suspension and refiltration, and reduction in copper-cadmium columns (A.P.H.A., 1971). Total-P was determined from additional unfiltered samples, also by the ascorbic acid method, but after predigestion with potassium persulphate and sulphuric acid (A.P.H.A., 1971).

A bathymetric map was also constructed for this lake from line soundings taken on theodolite bearings with electro-magnetic distance measurements. Hypsographic curves and other morphometric parameters were then calculated by planimetry from this map, following the methods of Wetzel and Likens (1979).

Because of water level fluctuations, all data for Lake Fidler and Sulphide Pool were plotted to an arbitary datum level, 0.70 metres on the gaugeboard in Lake Fidler, and 0.60 metres on that in Sulphide Pool. Similarily, all measurements from Lake Murchison have been referenced against a datum level of 225 metres above mean sealevel.

The heat content, thermal stability (for closed lakes - Walker, 1974), Birgean Wind Work, and volume weighed averages of temperature (and oxygen in some lakes) were calculated for all lakes using the computer program LIMNO/2 (Ferris, 1985). This is a modified version of the original LIMNO program of D.H. Merritt (Johnson *et al*, 1978). An initial density of 1 g cm^{-3} (pure water at 4°C) was assumed for wind work calculations.

Bowling (1981) found a highly significant linear relationship between conductivity at 18°C and density at 20°C [$\rho = 0.998244 + (4.867 \times 10^{-7})K_{18}$, $r^2 = 0.998$, $n = 92$] for samples from all three Gordon River meromictic lakes. This equation permits the calculation of chemical stabilities for these lakes for any occasion when a conductivity profile is available. Data of King (1980), King and Tyler (1982a, 1983), Baker *et al* (1985a) and unpublished, Bowling

(1981), and Croome (1984) were used, additional to that collected for this study, to calculate meromictic stabilities for the entire period the lakes have been studied, using the formula of Walker (1974) for an open lake.

5.3 RESULTS

PART A LAKE CHISHOLM

5.3.1 Physicochemical Features of Lake Chisholm

(a) Morphometric parameters

A bathymetric map of Lake Chisholm is presented in Figure 5.4, and derived morphometric parameters given in Figure 5.5 and Table 5.1. The lake is roughly circular, covers an area of almost 4.5 hectares, and has a maximum depth of 13.1 metres, although the water level varies considerably, being lowest in late summer and early autumn, and highest during spring (see Figures 5.6 and 5.8). The lake contains three small, steepsided basins with shallower ridges between them. 80% of the lake is less than 7.5 metres deep (Figure 5.5), and over 80% of the lake volume occupies the surface 5.5 metres. Despite this, Lake Chisholm still has a mean depth of 4.6 metres, and a relative depth of 5.5%, indicating that the maximum depth is considerable for a lake of its size. Lakes such as this are more likely to have retarded circulation and greater stability than large, shallow lakes (Wetzel and Likens, 1979).

(b) Water colour and light penetration

Surface g_{440} values of Lake Chisholm ranged between 24.0 m^{-1} and 31.5 m^{-1}, equivalent to 300-385 mg Pt L^{-1} on the Hazen scale (see Chapter Three). Water colour varied slightly both temporally and with depth (Figure 5.6), being least from midsummer to early winter, but increasing during the second half of the year. Colour was uniform with depth during periods of circulation, but usually increased with depth over most of the stratified period.

P.A.R. is very rapidly diminished with depth in the polyhumic waters of this lake, and the euphotic depth is extremely shallow, usually less than one metre below the surface (Figure 5.7). Mean downwelling vertical attenuation coefficients, $Kd_{(av)}$, were relatively high (Table 5.2), and downwelling P.A.R. also undergoes marked spectral modification to produce a predominantly red underwater light-field by

Table 5.1: Morphometric parameters of Lake Chisholm derived from the bathymetric map (Figure 5.4)

Elevation	120.0 m	Area	4.486 ha
Maximum length	290.0 m	Volume	207,340 m^3
Maximum width	238.0 m	Shoreline	886 m
Maximum depth	13.1 m	Shoreline development	1.2
Mean depth	4.6 m		
Relative depth	5.5%		

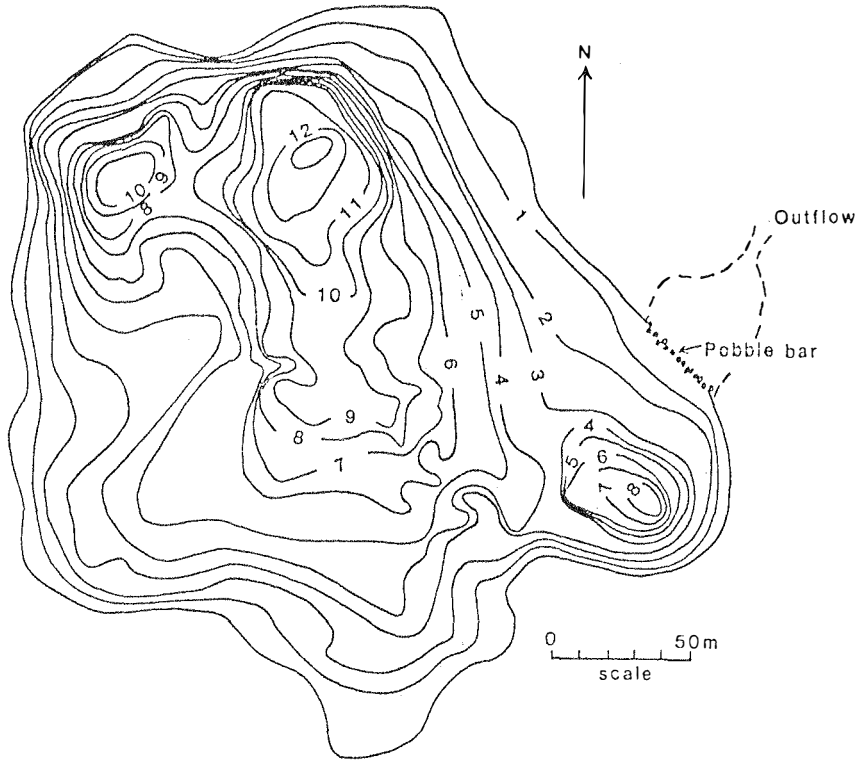


Figure 5.4 Bathymetric map of Lake Chisholm. Contours in metres.

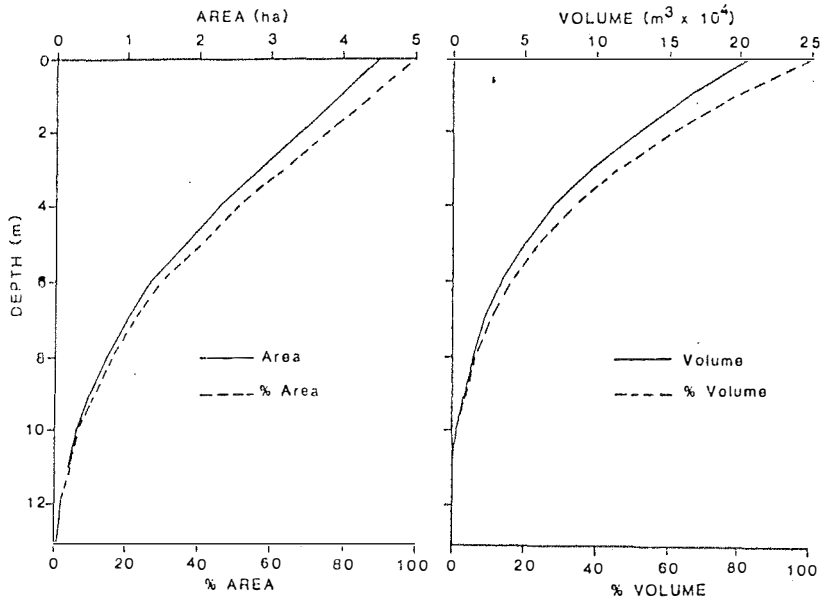


Figure 5.5 Depth-area and depth-volume relationships of Lake Chisholm.

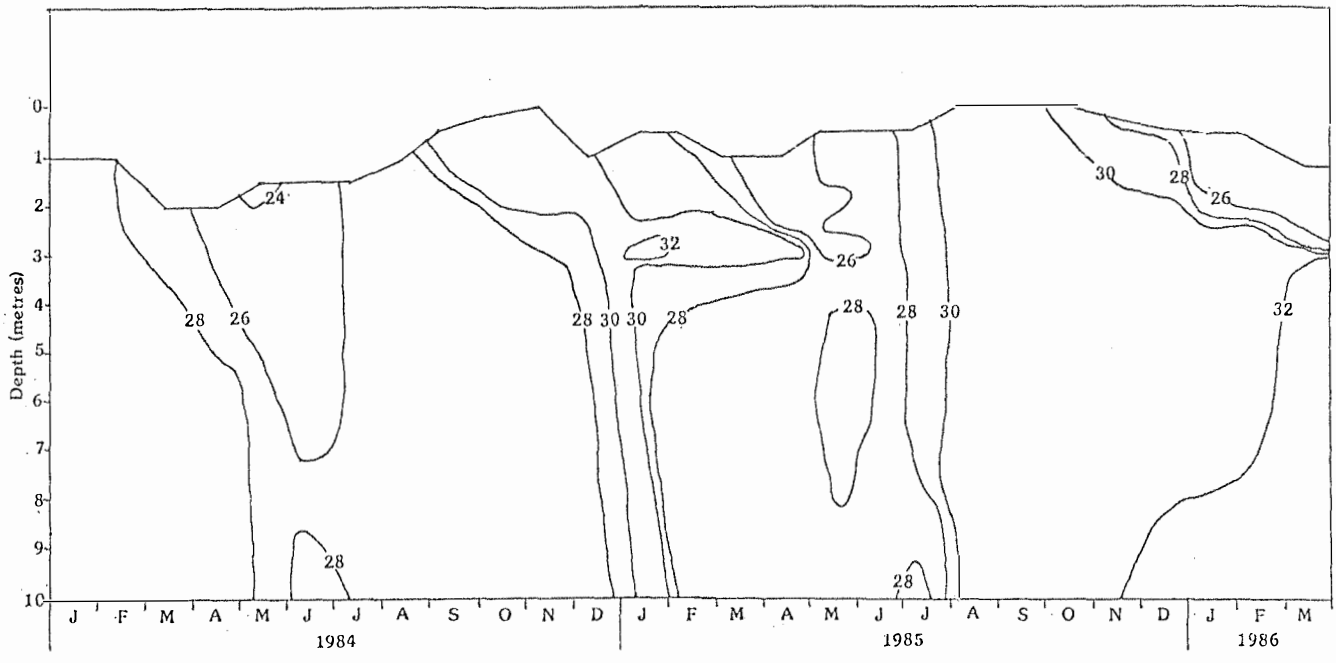


Figure 5.6 Isopleths of g_{440} in Lake Chisholm. Values in m^{-1} (ln units).

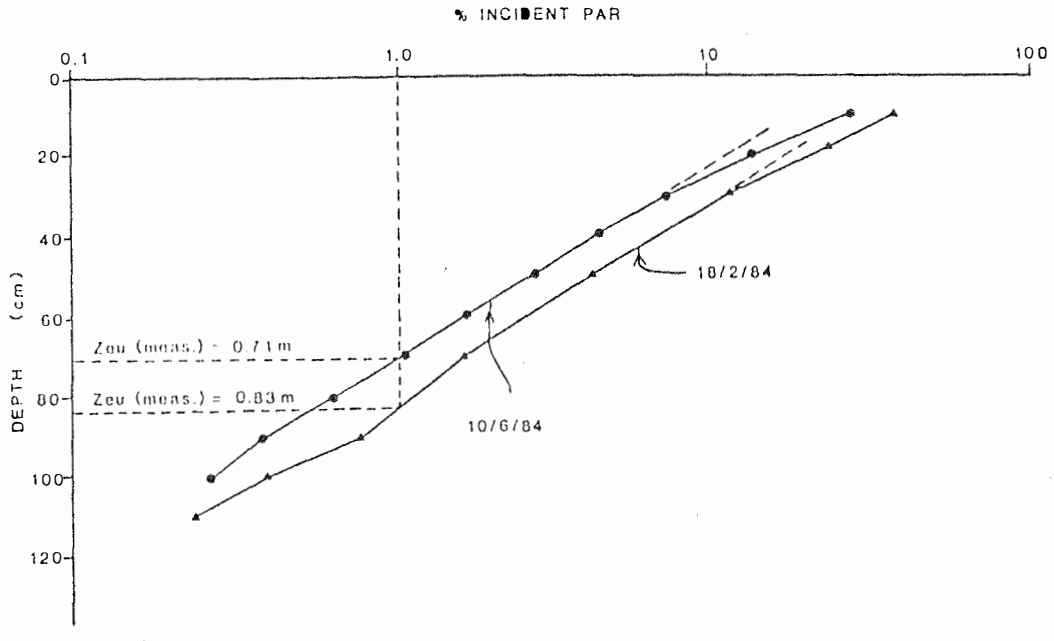


Figure 5.7 Selected profiles of downwelling P.A.R. (400-700 nm.) in Lake Chisholm, highlighting the rapid attenuation and shallow euphotic depth in the lake.

Table 5.2: Measured euphotic depths (Zeu) and mean vertical attenuation coefficients for downwelling P.A.R. ($Kd_{(av)}$) for Lake Chisholm

Date	Zeu (m)	$Kd_{(av)}$ (m^{-1})
18-2-84	0.80	4.88
16-4-84	0.90	4.51
10-6-84	0.70	5.37
13-8-84	1.00	3.82
9-9-84	0.80	4.60
7-10-84	0.80	4.34
6-11-84	0.80	5.10
11-1-85	0.80	4.80
6-2-85	0.70	5.10
4-8-85	0.75	4.45
22-3-86	0.75	5.15

0.5 metres deep (see Chapter Three). Turbidity has little effect as values were always below 1.0 N.T.U., and usually below 0.5 N.T.U. in the surface waters. Secchi disc depths ranged from 0.8 to 1.2 metres.

(c) Thermal stratification, dissolved oxygen, and dissolved sulphides

The attenuation of most solar energy close to the surface has a considerable effect on the thermal stratification of Lake Chisholm. Isopleths of temperature (Figure 5.8) show the lake to be warm monomictic, by Lewis's (1983) classification. Stratification commences early in spring (September), and cooling does not commence until mid autumn, with overturn only occurring in late May, so that the period of winter circulation is three months or less. The isopleths and selected profiles (Figure 5.9) highlight the intense shallow thermal gradients present during the summer months, which span a temperature difference of 10-12°C, with an epilimnion only one to two metres deep.

The monthly volume weighed average lake temperatures and heat contents (Figure 5.10a) reveal the lake's annual heating cycle. These closely follow the local climatic conditions (Figure 5.2), being lowest during July and August, but rise rapidly in response to warming air temperatures and increasing solar radiation in spring, reaching a maximum during mid-summer before declining again throughout autumn. There were only slight variations between the maximum and minimum average temperatures and heat contents of the two study years. The annual heat budget, θ_{ba}, for the 1984-85 heating period was 2410 cal cm^{-2}; that for 1985-1986 2124 cal cm^{-2}.

The yearly variation in dissolved oxygen and sulphides is shown in Figure 5.11, along with some monthly profiles in Figure 5.9. These closely follow the regular cycle of the thermal regime. The onset of hypolimnetic anoxia lags behind the onset of thermal stratification, with small amounts of oxygen being present to the sediments into November. However the hypolimnion is anoxic, with considerable amounts of dissolved sulphides present, for six to seven months of the year. Oxic waters descend with the deepening thermocline in late autumn, but the lake is not oxygenated throughout its entire depth until after overturn, in June.

The oxygen content in Lake Chisholm rarely exceeded 75% of the saturation value, even in the surface waters. This decreases even further in winter when anoxic bottom waters mix with the oxygenated

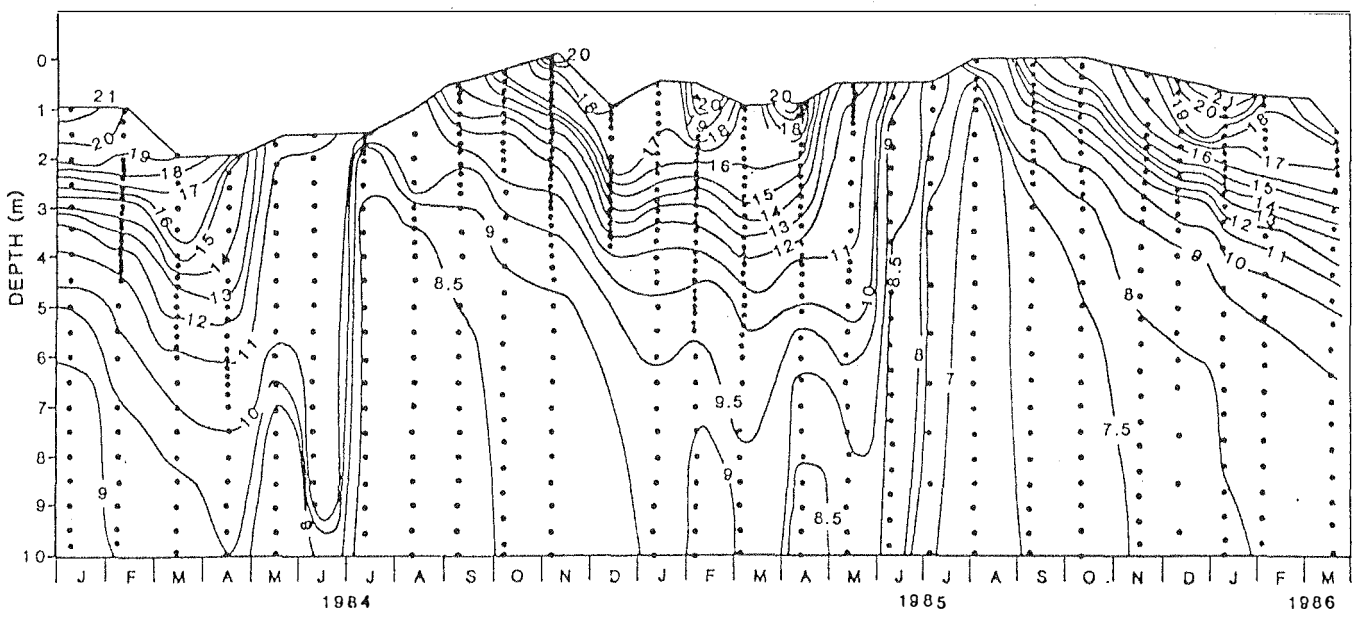


Figure 5.8 Isotherms (°C), showing the warm monomictic nature of Lake Chisholm.

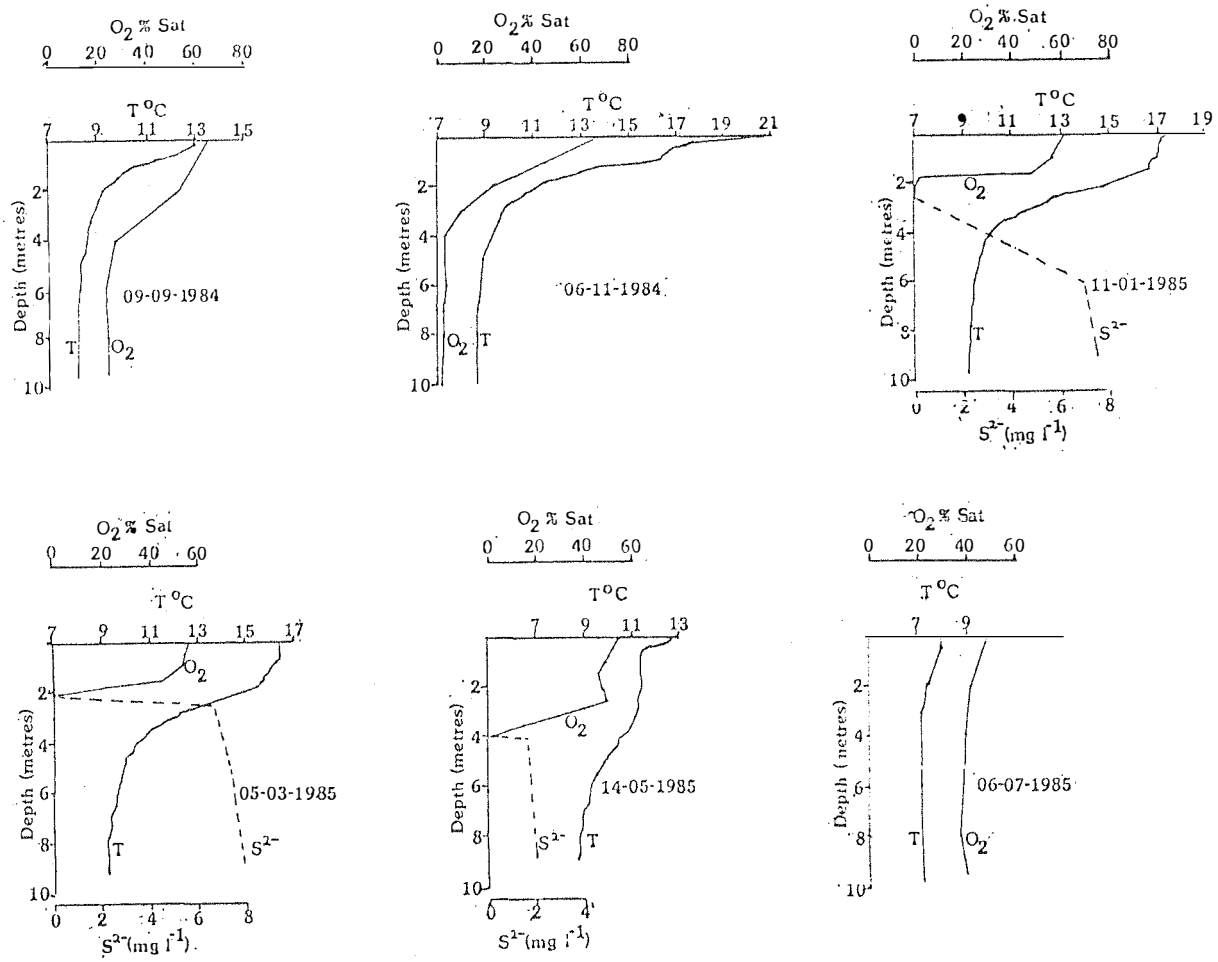


Figure 5.9 Thermal, oxygen, and dissolved sulphide profiles, at two monthly intervals, for Lake Chisholm. These highlight the annual stratification cycle of the lake.

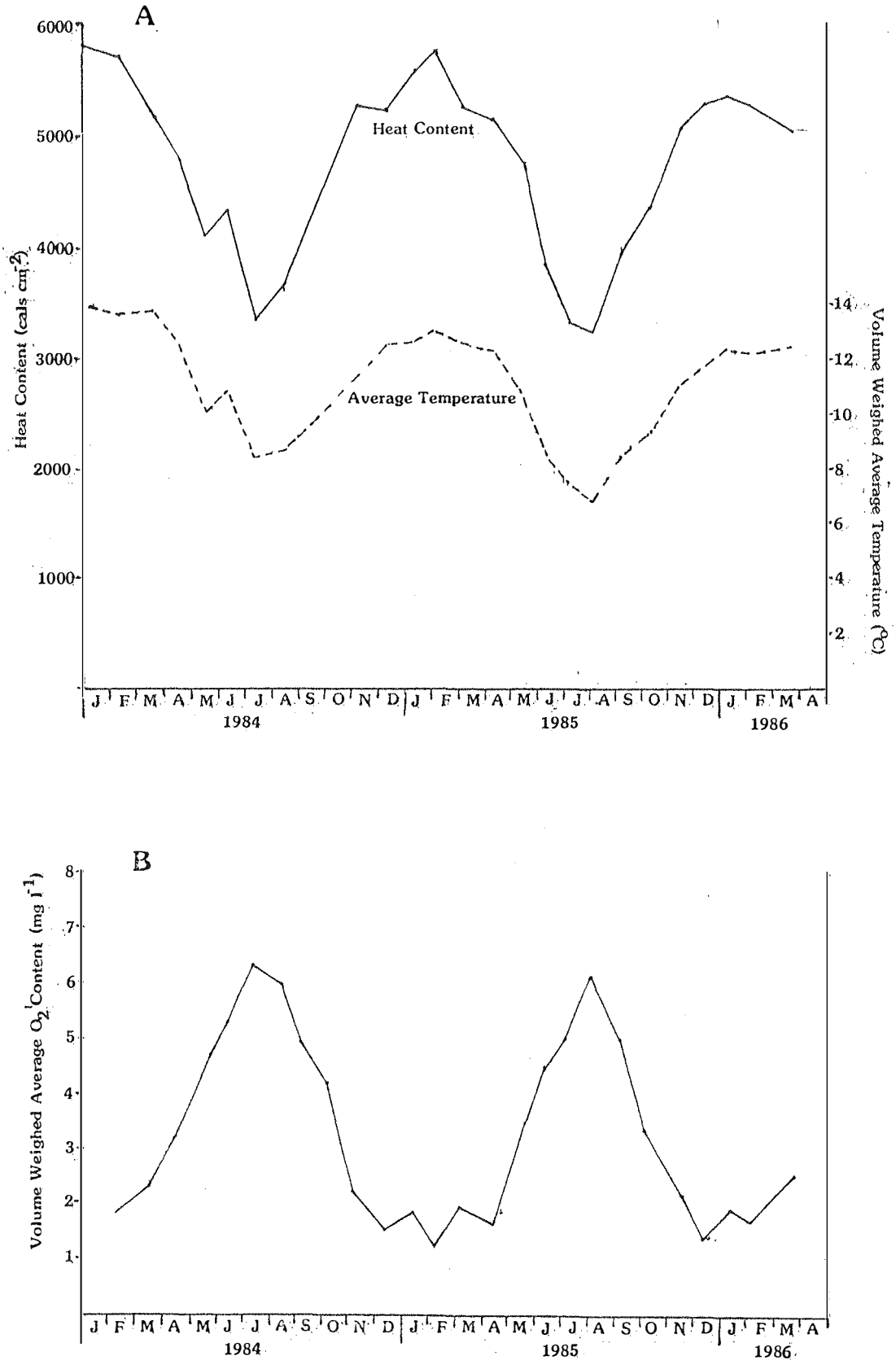


Figure 5.10 A. Monthly values of heat content (solid line) (cals cm^{-2}) and volume weighed average temperature (dashed line) (°C) for Lake Chisholm. B. Those for volume weighed average oxygen content (mg l^{-1}).

surface layers. Because a large volume of the lake is occupied by the anoxic hypolimnion during summer, the whole lake average for oxygen falls below 2 mg L^{-1}, but the maximum value was still only a little above 6 mg L^{-1} during winter, when circulation was at its greatest extent (Figure 5.10b). This is less than 55% of the saturation possible at the average lake temperature at the time.

(d) Apparent redox potentials (Eh), pH, and conductivity (K_{18})

Isopleths for apparent redox potentials (Eh) are shown in Figure 5.12. These closely follow those for oxygen. Summer surface values were around 400 mv., but these fell rapidly to 40 mv. below the thermocline, due to the reducing nature of the anoxic, hypolimnetic waters. During winter values remain at 400 mv. at all depths. Due to equipment breakage, little data were available for 1985.

pH varied between 4.8 and 5.8 throughout the study period, with values usually between 5.2 and 5.6 (Figure 5.13). pH was lowest in spring and early summer each year, but changed by as much as 0.8 of a unit below the thermocline. Sometimes this change represented an increase in pH, but at other times pH decreased. During periods of circulation pH was constant with depth.

There was only slight vertical variation in the concentrations of ionic solutes during the study, as indicated by the isopleths for electrical conductivity (Figure 5.14). K_{18} was usually close to 100 $\mu S\ cm^{-1}$, but surface values tended to increase slightly during periods of maximum thermal stratification, when any slight density increases due to small increases in solutes would be counteracted by the decreased density of the warmer water. The cells of surface water of higher than normal conductivity appear to be inflows which depressed the thermocline and oxycline to slightly deeper levels.

(e) Major ions, iron, and manganese

Few differences were shown in the ionic characteristics of Lake Chisholm waters, either in depth or time. The distributions of the major ions are given in the ternary diagrams of Figure 5.15. Despite the karstic nature of the lake's formation, the water chemistry tends towards dilute sea water, being dominated by sodium and chloride, with little enrichment from alkaline earth bicarbonates.

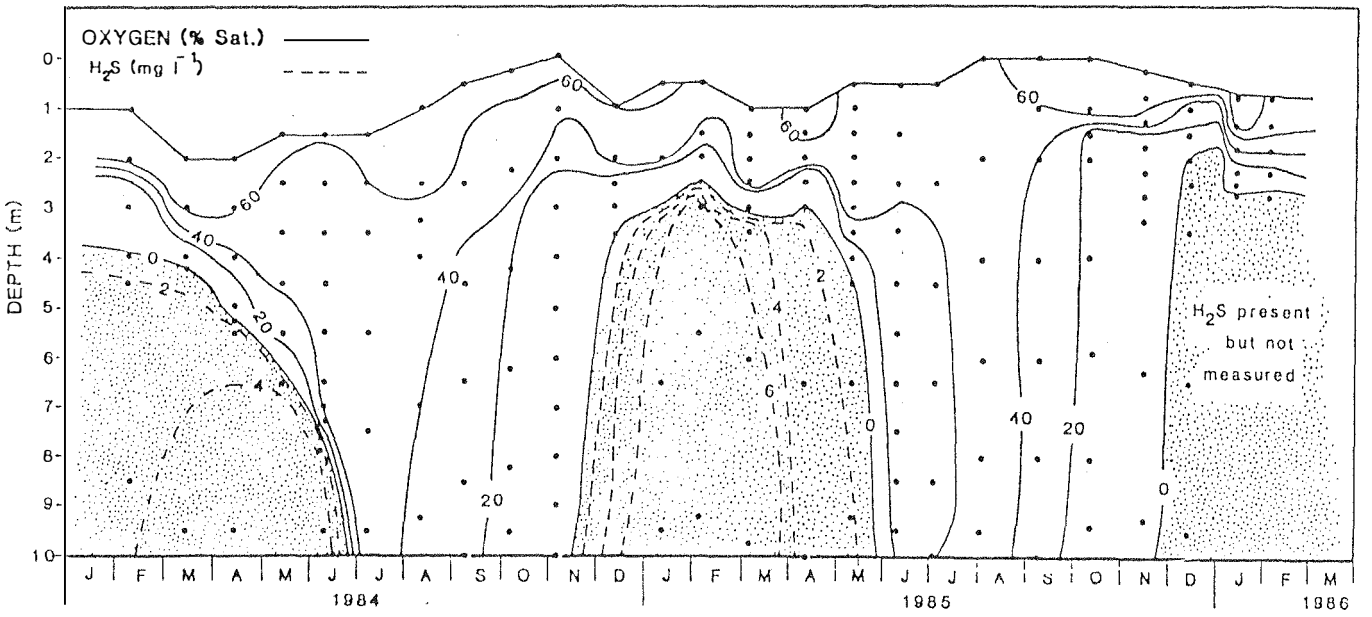


Figure 5.11 Isopleths of dissolved oxygen (% saturation) and total dissolved sulphides (mg l^{-1}) in Lake Chisholm. The stippled areas indicate anoxia and the presence of H_2S.

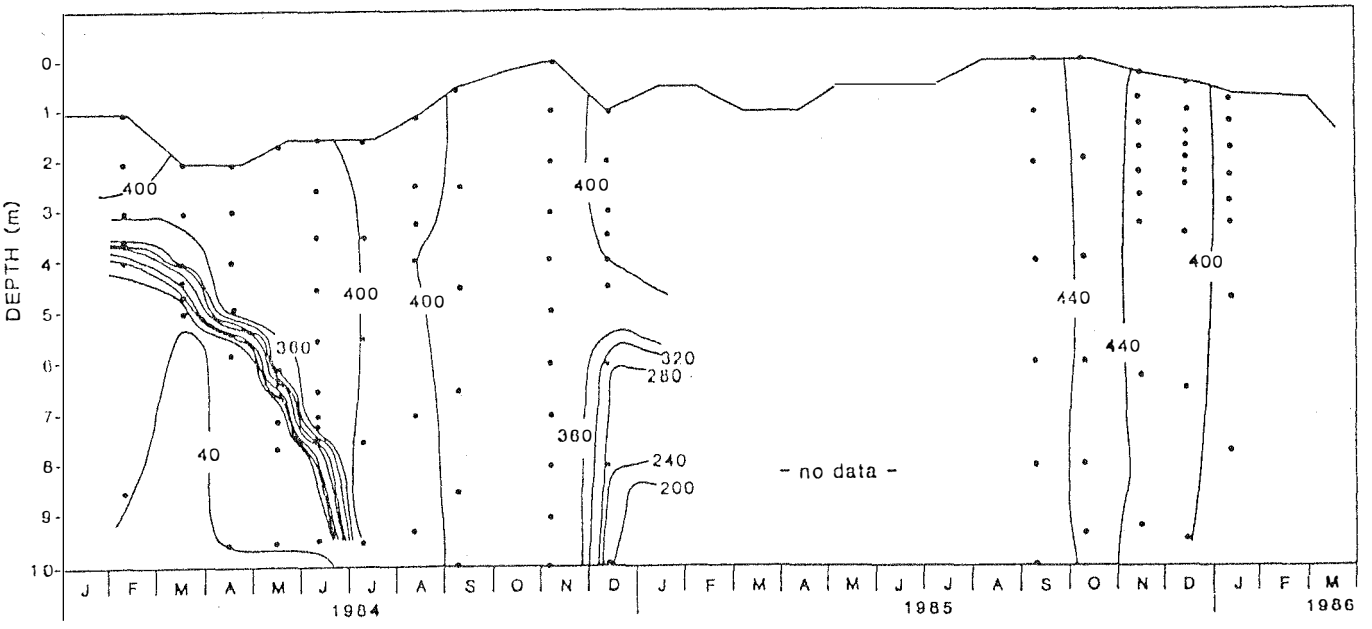


Figure 5.12 Isopleths of apparent redox potential, Eh (mv, not corrected for pH) in Lake Chisholm.

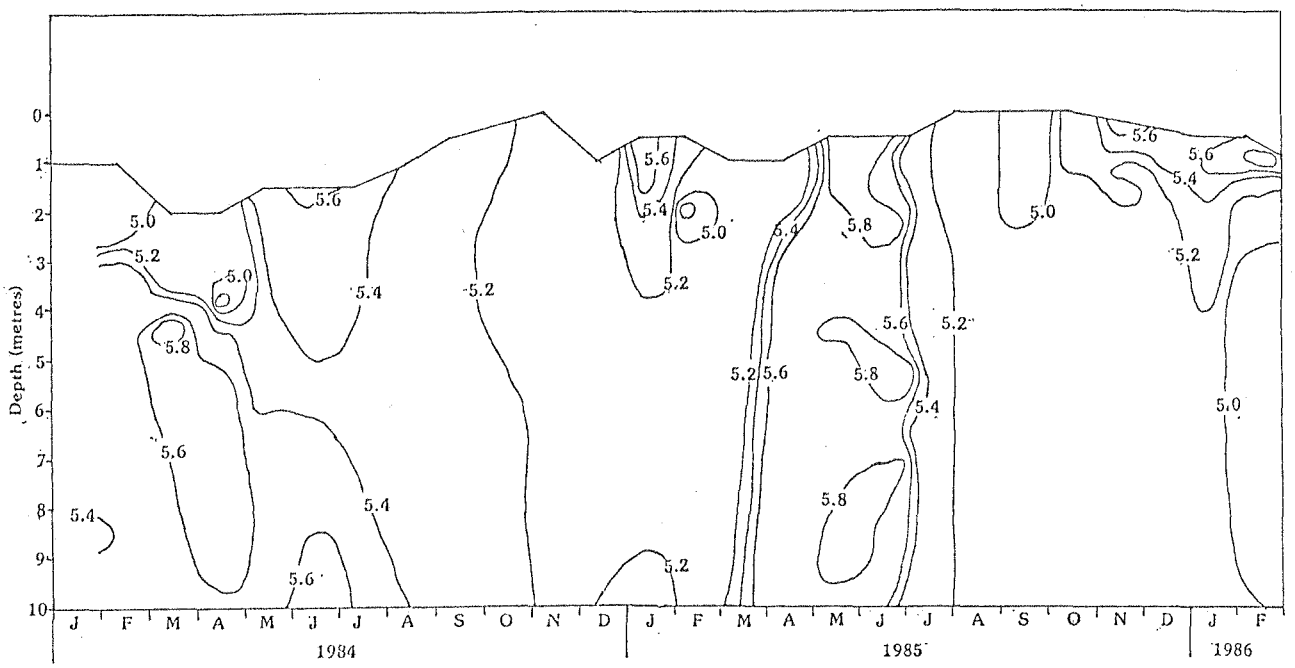


Figure 5.13 Isopleths of pH in Lake Chisholm.

Figure 5.14 Isopleths of electrical conductivity at 18°C (K_{18}) (μS cm^{-1}) in Lake Chisholm.

Iron never exceeded 0.2 mg L^{-1} in the surface waters, and 0.5 mg L^{-1} in the hypolimnion; while manganese was not measurable, even at the height of thermal stratification and hypolimnetic anoxia.

(f) Nutrient chemistry

Both NO_3-N (including NO_2-N) and Total-P were uniformly distributed with depth during circulation in August, but there is evidence of a modest accumulation of nutrients in hypolimnetic waters during the stratified period from September onwards (Table 5.3). Epilimnetic nitrate levels also decreased following the onset of thermal stratification, as did levels of Total-P. Orthophosphate forms about 50% of the Total-P in Lake Chisholm.

5.3.2 Thermal Stabilities and Birgean Wind Work for Lake Chisholm

The monthly values of thermal stability are plotted in Figure 5.16. Annual change in thermal stability closely resembles those of heat content and average temperature (see Figure 5.10a), with a summer maximum of 50 to 60 gm-cm cm^{-2}, and winter minimums below 2 gm-cm cm^{-2}. The marked dips in stability that occurred in January and March, 1985, correspond to times of considerable cooling of the surface waters and deepening of the thermocline, caused by possible inflows (see Figure 5.8 and Section 5.3.1(d), above).

Birgean Wind Work (Figure 5.16) also follows the monthly patterns set by heat content, average temperature, and stability. The maximum values declined over the three successive summers, and winter values in 1985 were markedly lower than those of midwinter, 1984, resulting from the cooler average temperatures of the lake during 1985. The unusual marked increase in June, 1984, corresponds with a rapid deepening of the thermocline just prior to overturn, at that time.

Direct monthly work curves show exactly where energy has been expended in the water column (Johnson et al, 1978). Examples of these, at three-monthly periods during 1985, are given in Figure 5.17, and shown as isopleths for the whole study period in Figure 5.18. Most energy exerted on the lake to achieve the observed thermal gradients is distributed in the surface 2-3 metres. Only in winter during circulation is energy distributed more or less evenly throughout the water column.

Table 5.3: The distribution of nutrients ($\mu g\ L^{-1}$) in Lake Chisholm in 1985. N = NO_3-N + NO_2-N; P = PO_4-P; TP = Total-P

Depth (m)	August			September			October			November			December		
	N	P	TP	N	P	TP	N	P	TP	N	P	TP	N	P	TP
0	56		62	58	0	31	10	18	28	13	15	38	10	53	40
1.0	-		-	-	-	-	-	-	-	-	-	-	6	35	46
2.0	50		53	65	2	52	11	26	46	4	32	57	5	49	48
4.0	-		57	106	5	46	34	31	48	44	32	73	17	53	55
6.0	57		57	95	17	45	-	30	61	68	35	53	-	-	-
7.0	-		-	-	-	-	-	-	-	-	-	-	-	48	62
8.0	65		53	68	5	50	40	35	49	70	39	60	-	-	-
bottom	58		63	176	4	65	43	33	58	63	42	88	20	49	53

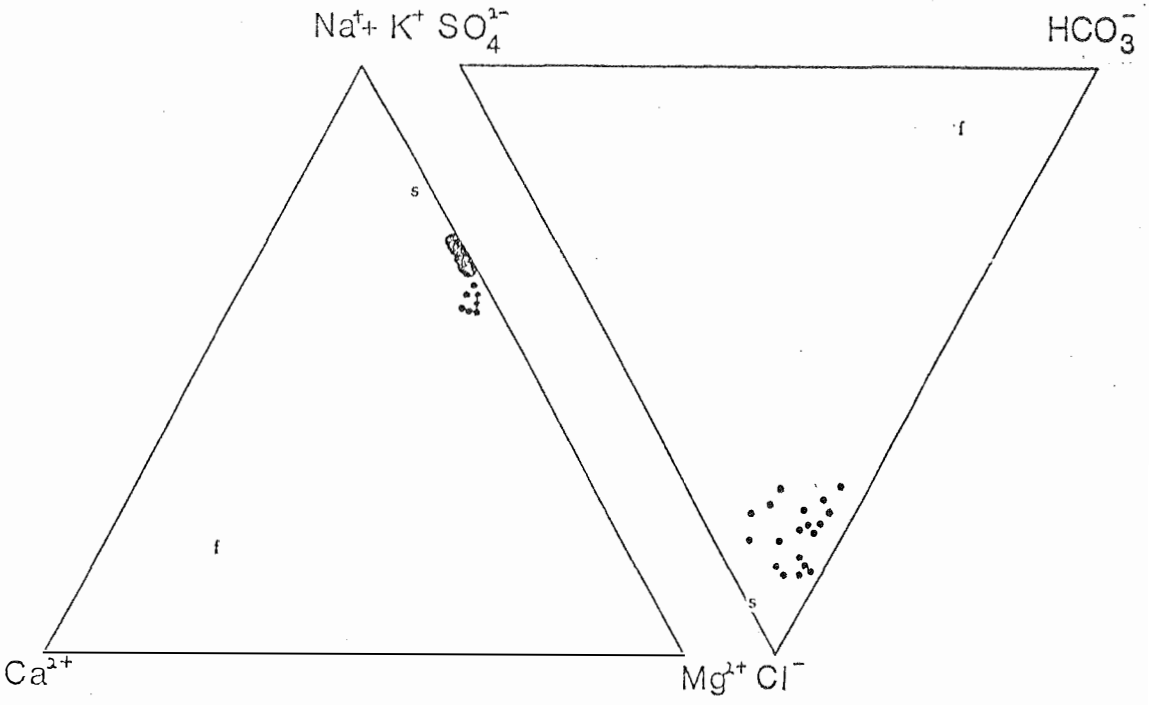


Figure 5.15 Ternary diagrams of ionic composition of Lake Chisholm waters from various depths and times. f = World Average Freshwater, s = seawater.

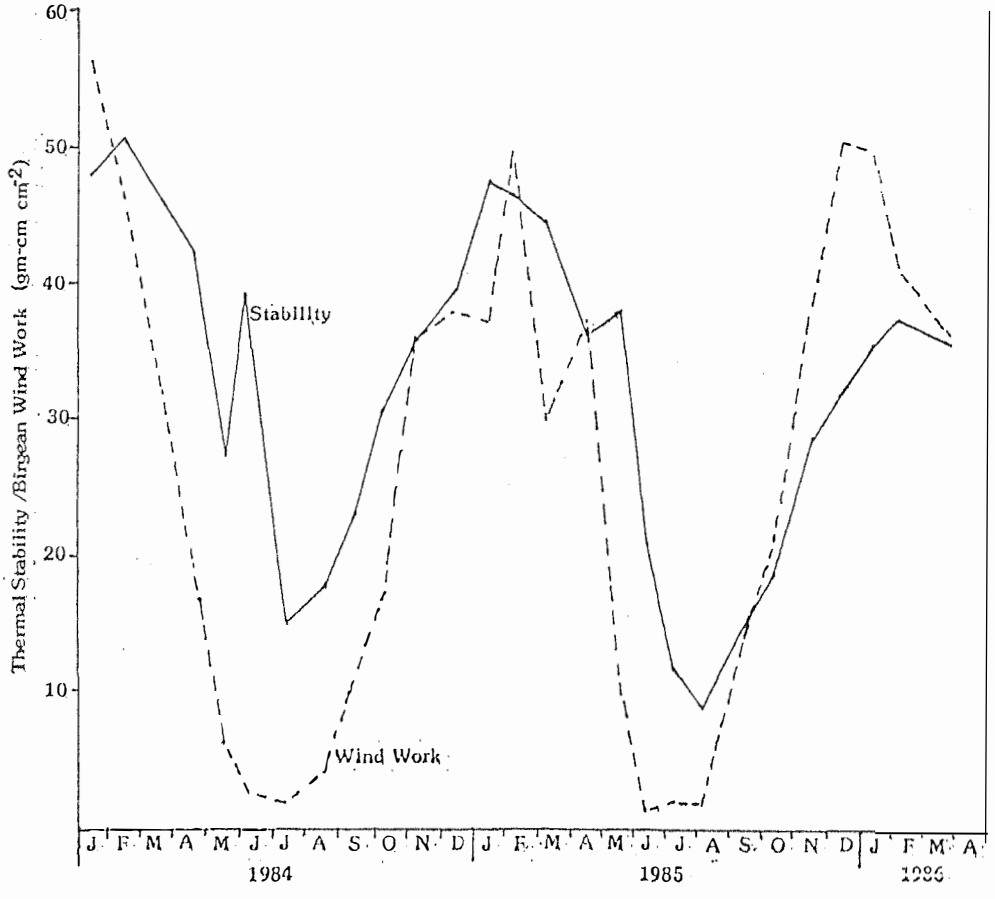


Figure 5.16 Monthly values of thermal stability (solid line) and Birgean Wind Work (dashed line) (both gm-cm cm^{-2}) for Lake Chisholm.

Differential work curves (ΔB, the direct work curve for one month minus that of the previous month - Johnson *et al.*, 1978) are shown in Figure 5.17. Positive work ($\Delta B > 0$) is done over most depths of the lake during spring and summer as the lake waters are warmed, but the curves become negative ($\Delta B < 0$) during autumn and winter as the lake cools. Figure 5.19 gives isopleth details of ΔB, and highlights three features. Firstly, the lake undergoes extensive periods during the year when positive work is acting to distribute energy throughout the lake, but only short periods of negative work exist, when the lake looses heat to the surrounding environment. These brief cooling periods occur mainly in winter, but even then can be interrupted by periods when positive work again warms the lake. Secondly, in autumn, although negative work causes cooling to take place close to the surface, deeper within the lake positive work is still occurring as cooler hypolimnetic water is replaced by the warmer overlying waters of the sinking thermocline. The third, and most significant feature is that most of the deeper waters experience very little work, either positive or negative, acting upon them during the course of a year, and those closest to the bottom are obstensibly stagnant (when $\Delta B = 0$) for much of the time.

5.3.3. Biological Features of Lake Chisholm

Tows using a 20 µm plankton net indicate that flagellates are the dominant phytoplankton of Lake Chisholm, although some species, especially those of small size, may have been missed by this sampling method. A number of taxa of Chrysophyceae occurred as virtually monospecific blooms on occasions. Several species of *Mallomonas*, especially *M. morrisonensis* Croome and Tyler, and a colonial green flagellate, possibly *Gonium sociale* (Duj.) Warming, were abundant in the late summer and early autumn of 1984. These had disappeared by May, when the non-motile green alga *Crucigenia quadrata* Morren was very plentiful in the epilimnion, and the few flagellates present were mainly *Peridinium* sp. A bloom of *Synura mamillosa* Takahaski occurred in August, 1984, while the following month *Peridinium* sp. was the dominant flagellate, with some *S. mamillosa* and *Dynobryon cylindricum* Imhoff also being present.

Chrysophytes dominated the phytoplankton for the remainder of the study. Blooms of *M. morrisonensis* occurred in November, 1984; of *D. cylindricum* in October 1985 and January, 1986; while *M. splendens* (G.S. West) Playfair were present in large numbers in December, 1984; and from May to July of 1985 . Attendant species recorded in small numbers either occasionally or constantly included *M. adamas* Harris and Bradley em.

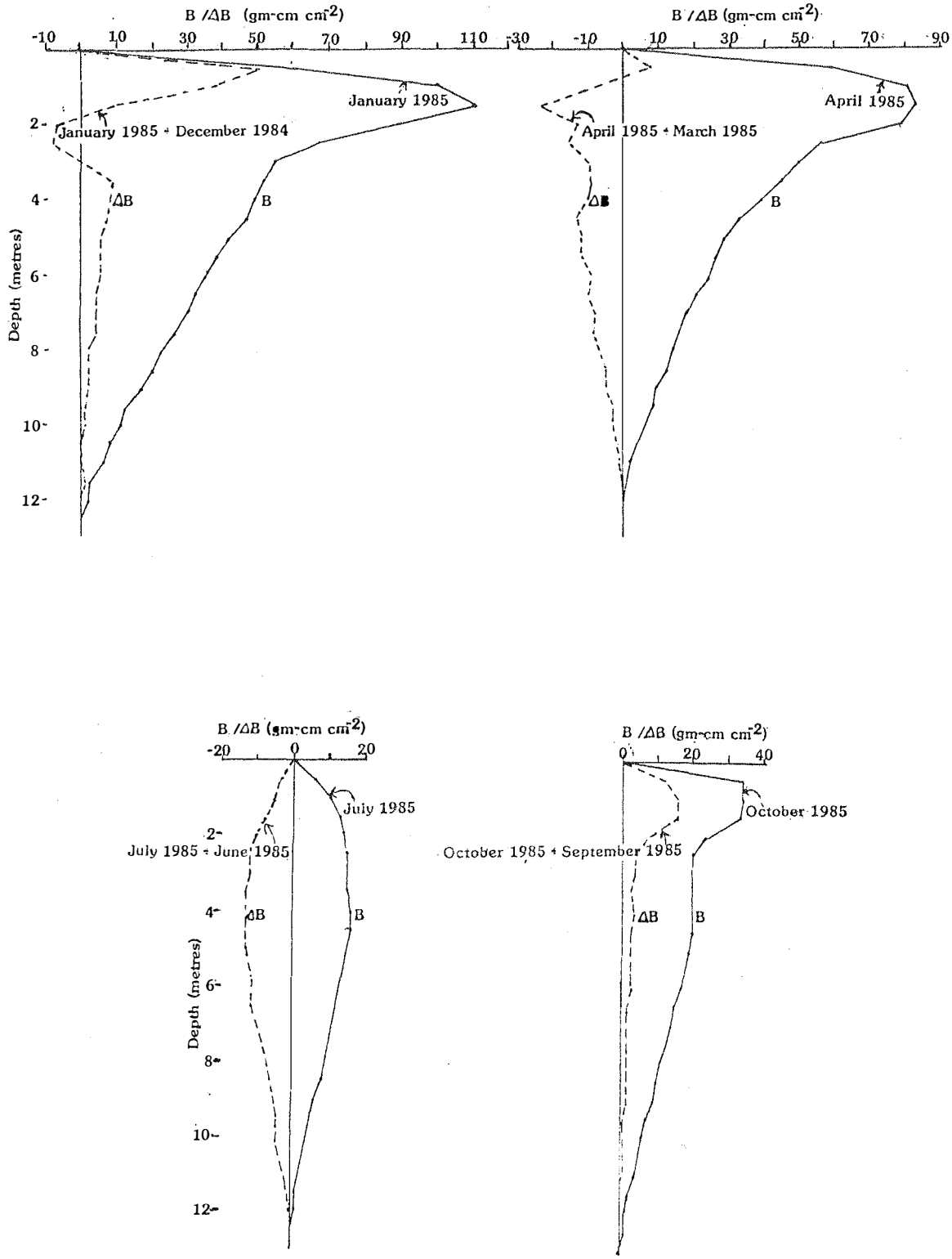


Figure 5.17 Direct work curves (B) (solid lines), and differential work curves (Δ B) in summer, autumn, winter, and spring, for Lake Chisholm in 1985. B values × 10^{-2}

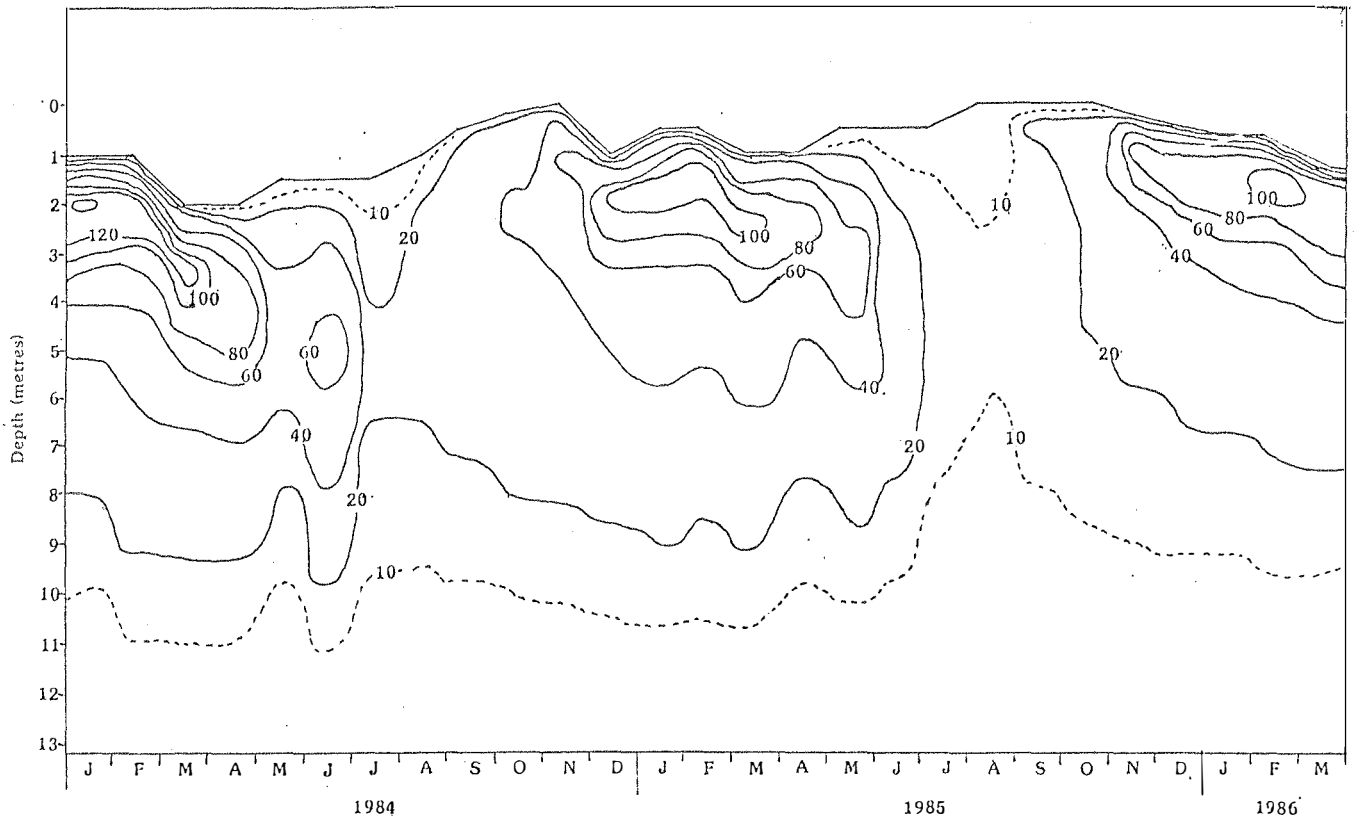


Figure 5.18 Isopleths of direct work (B) (gm-cm cm^{-2}) done by wind to distribute energy within Lake Chisholm. B values $\times 10^{-1}$

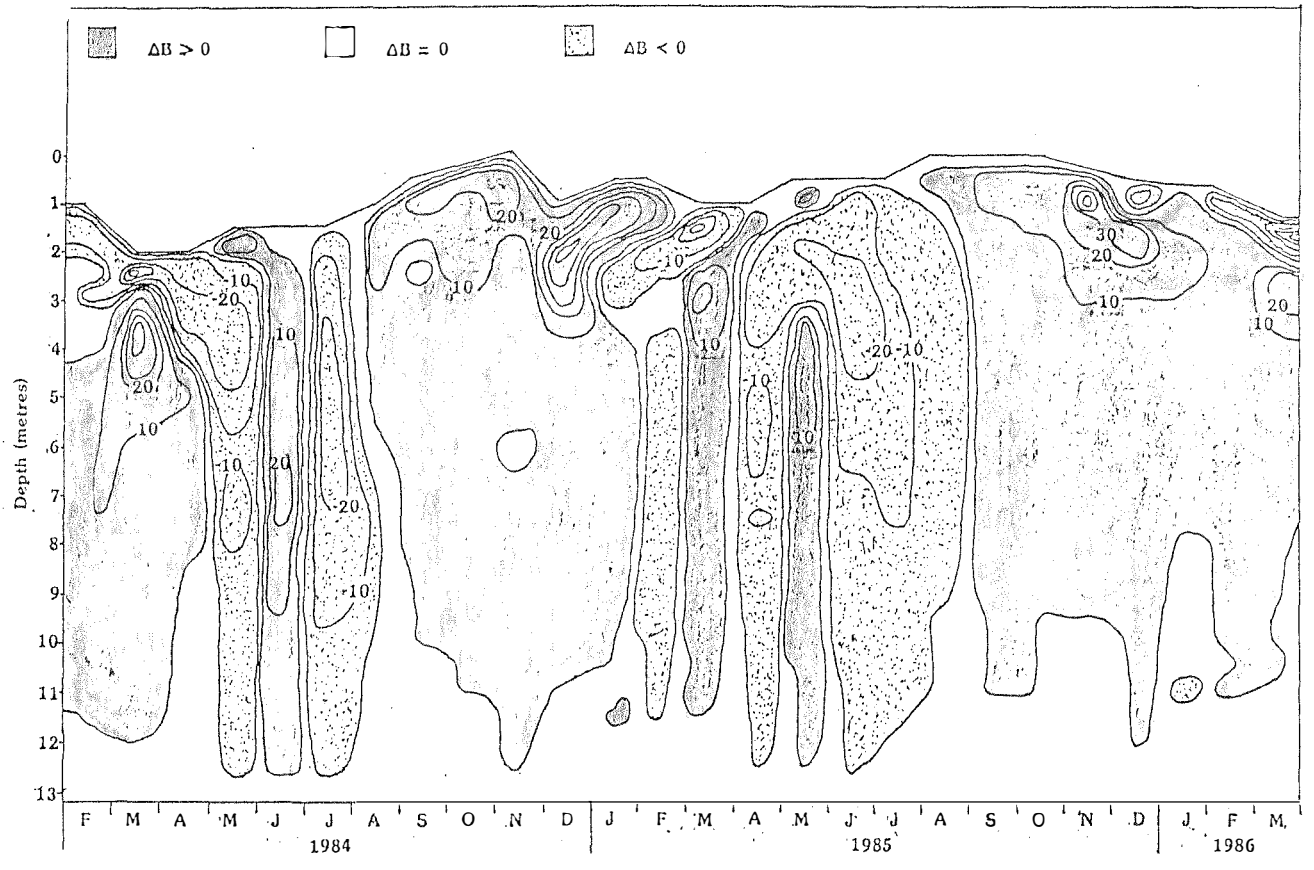


Figure 5.19 Differential work (Δ B) isopleths (gm-cm cm^{-2}) for Lake Chisholm. B values $\times 10^{-1}$

Dürrschmidt and Croome; *M. tasmanica* (Croome and Tyler) Asmund and Kristiansen, *Merotrichia bacillata* Mereschkowsky, *Prorocentrum playfairi* Croome and Tyler, *Botryococcus braunii* Kützing, and *Anabaena* sp. Desmids were unusual in the lake, the most common being *Cosmarium* sp., and diatoms were rare.

Zooplankton comprised mainly of a calanoid copepod (probably *Calamoecia tasmanica* (Smith)), and several species of rotifers were also common. *Chaoborus* sp. larvae, notonectids, and tadpoles (species unknown) were also very common during summer. There was no evidence of fish in the lake, which however supports a small population of duckbilled platypus (*Ornithorhynchus anatinus* Shaw and Nodder). The lake and its surrounding forest also provides a habitat for several species of waterbirds.

PART B: THE RESERVOIRS OF THE PIEMAN RIVER, AND LAKE BARRINGTON

5.3.4. Physicochemical Features of Lake Mackintosh

Lake Mackintosh is humic and non-turbid. Surface gilvin values were between 7.8 and 8.3 m^{-1}, while turbidity was around 0.5 N.T.U. Both colour and turbidity increased dramatically with depth in March, 1981, (Hydrology Section, H.E.C., unpublished), although this may be due to iron and manganese rather than to dissolved humus (see Section 5.3.5(a), below).

Chemical stratification had developed by March, 1981, six months after the sealing of the dam (Table 5.4). The lake was thermally stratified and anoxic below twelve metres, while conductivity, magnesium, potassium, bicarbonate, iron, manganese, and dissolved sulphides all increased below this depth. Calcium and bicarbonate were the dominant ions, apart from in the surface waters.

The chemical stratification had weakened considerably by June, 1981, and occupied only the bottom few metres (Table 5.4). It had disappeared entirely by August, 1981, when small amounts of oxygen were present at the bottom, and sodium and chloride were the dominant ions throughout. Subsequent data collected in 1982 and 1983 (Table 5.4 and Figure 5.20) indicate that chemical stratification did not redevelop. In November, 1983, there were no significant changes in

Table 5.4: Chemical data for Lake Mackintosh for the first year of impoundment (1981-82) (data from Hydrology Section, Hydro-Electric Commission of Tasmania, unpublished)

Depth	Maximum Depth	Temp.	Oxygen	K_{18}	Ca^{++}	Mg^{++}	K^{+}	HCO_3^{-}	Fe	Mn	$S^{=}$
(m)	(m)	(°C)	(%Sat.)	(μScm^{-1})	meq L^{-1}	meq L^{-1}	meq L^{-1}	meq L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}
					11-03-81						
3	49	17.8	68.4	39.8	0.150	0.083	0.017	0.125	0.14	0.033	< 0.10
12		12.0	0.9	40.5	0.185	0.083	0.016	0.161	0.56	0.060	0.39
43.5		8.0	0.0	57.7	0.305	0.165	0.046	0.262	8.10	0.630	0.40
48		7.9	0.0	59.6	0.325	0.174	0.050	0.269	8.50	0.730	0.36
					10-06-81						
3	43	10.7	63.1	37.1	0.140	0.083	0.018	0.138	0.27	0.027	< 0.05
12		10.3	44.6	38.1	0.135	0.083	0.024	0.144	0.34	0.028	< 0.05
38		8.4	0.0	39.0	0.155	0.091	0.017	0.197	0.81	0.043	< 0.05
42.5		8.4	2.6	46.7	0.250	0.107	0.018	0.230	1.73	0.181	0.25
					20-08-81						
3	50	8.2	50.1	37.1	0.125	0.083	0.021	0.092	0.34	0.025	0.16
12		8.2	45.8	37.1	0.125	0.083	0.022	0.105	0.32	0.027	0.16
45.5		6.8	4.9	37.1	0.140	0.074	0.016	0.105	1.09	0.085	0.03
49.5		6.5	2.4	37.1	0.140	0.074	0.016	0.085	0.99	0.059	0.16
					20-04-82						
3	39.1	14.2	74.6	40.0	0.145	0.083	0.026	0.164	0.15	1.130	< 0.08
12		14.0	54.5	60.0	0.155	0.124	0.029	0.131	1.07	1.226	< 0.08
34.5		7.1	0	38.1	0.135	0.074	0.027	0.197	1.08	1.158	0.20
38.5		7.0	0	38.1	0.135	0.074	0.024	0.197	1.12	1.135	0.30

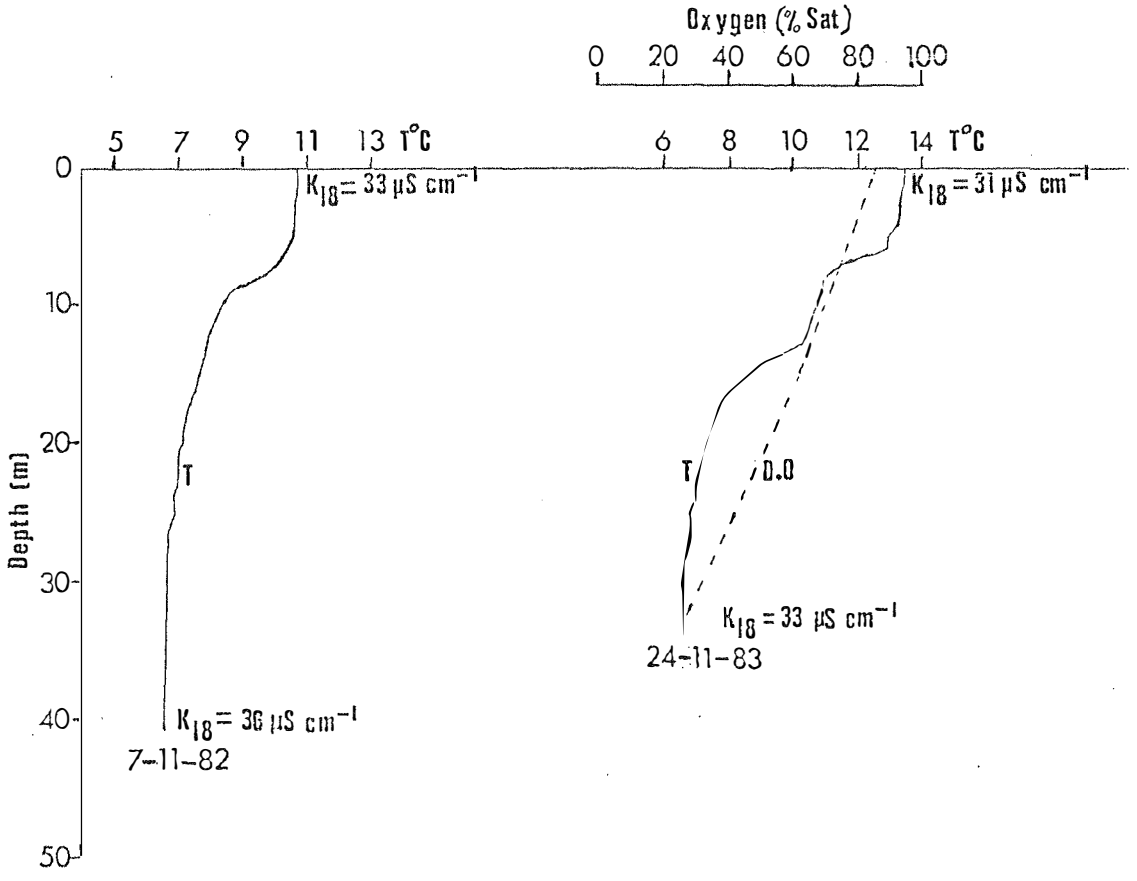


Figure 5.20 Temperature and dissolved oxygen profiles from Lake Mackintosh. Conductivity is also indicated.

K_{18}, concentrations of the major ions, or of iron and manganese between the surface and the bottom waters.

5.3.5 Physicochemical Features of Lake Murchison

(a) Colour and Turbidity

Lake Murchison surface waters are also humic and non-turbid. Gilvin ranged from 4.1 to 8.9 m^{-1}, and turbidity was generally below 1.0 N.T.U. Both tended to increase with depth, especially near the bottom of the reservoir, where a localized monimolimnetic pool was present. However such data must be treated with caution. Both dissolved and fine precipitates (those not adequately filtered out, even with a 0.22 μm membrane filter) of iron and manganese considerably increases measurements of water colour and turbidity, making such measurements highly suspect.

(b) Temperature, Oxygen, and Dissolved Sulphides

Thermal, oxygen, and dissolved sulphide profiles are shown in Figure 5.21. Initial investigations four months after the creation of the reservoir, in November, 1982, revealed a slight thermal stratification, and oxygen present close to the bottom. A possible warming of 0.1°C below 45 metres datum is insufficient indication of the onset of chemical stratification. By March, 1983, the lake was strongly stratified thermally, with temperature declining rapidly through the surface ten metres, followed by a more gradual decline to almost 35 metres datum. A definite increase of 0.2°C occurred over the bottom 20 metres. Oxygen values were low near the surface, but increased in the mid-depths of the reservoir, before falling to zero by 40 metres datum. Hydrogen sulphide was smelt in waters below this depth. Results from July, 1983, when thermal stratification was absent, confirmed the existance of a stable monimolimnetic pool. Considerable mixing is apparent throughout the upper 50 metres of the reservoir, but below this water temperature suddenly increased by more than 1.0°C, oxygen fell from 64% saturation to zero in the space of two metres, and hydrogen sulphide was present.

Marked dichothermal temperature profiles, with concomitant rapid changes from oxic to anoxic, sulpheretted waters were present over the next two years (Figure 5.21). A yearly trend is apparent, with the bottom of the oxycline and the thermal inversion increasing in depth, relative to datum, between March and November, during winter

circulation, but becoming shallower again over summer when thermal stratification was present. In July, 1985, oxygen at almost 20% saturation was measured near the bottom, but dissolved sulphides were again present at these depths the following November. Dichothermy persisted throughout.

(c) Redox potentials (Eh), and pH

Apparent redox potentials (Eh) further highlight the rapid change from oxic waters throughout most depths of the reservoir to the anoxic, reducing, sulphide laden waters of the monimolimnetic pool (Figure 5.22).

pH values were usually between 5.0 to 6.0 throughout most of the water column, but gradually increased towards 7.0 within the monimolimnetic pool (Figure 5.22). The magnitude of this pH change was greatest in the first half of the study.

The positions of both the pH and redoxclines also varied during each year, in a similar manner to the thermal inversions and the onset of anoxia (see Section 5.3.5 (b), above).

(d) Electrical Conductivity

The presence of chemical stratification in Lake Murchison is highlighted by the electrical conductivity measurements in Figure 5.23. K_{18} increased slightly from 25 to 37 $\mu S\ cm^{-1}$ between the surface and 50 metres datum, in November, 1982, and this may indicate the onset of chemical stratification. The magnitude of the K_{18} differences with depth had increased by March, 1983, in particular from below 35 metres datum. Subsequent data for 1983 and 1984 show a well developed, sharp, conductivity cline occurring near the bottom of the reservoir, at depths identical to those of the temperature increases, and of the redox changes. A maximum K_{18} value of 303 $\mu S\ cm^{-1}$ was measured from the bottom waters in November, 1983, but conductivity subsequently declined, and the cline had sunk to below 60 metres datum, by the first half of 1985. In July, 1985, there was little difference in K_{18} between the surface and the bottom waters, although by November, conductivity at 60 metres datum had again increased slightly.

Part of the variation seen in conductivity (Figure 5.23) (and in the major ions - see below) close to the bottom of the reservoir must

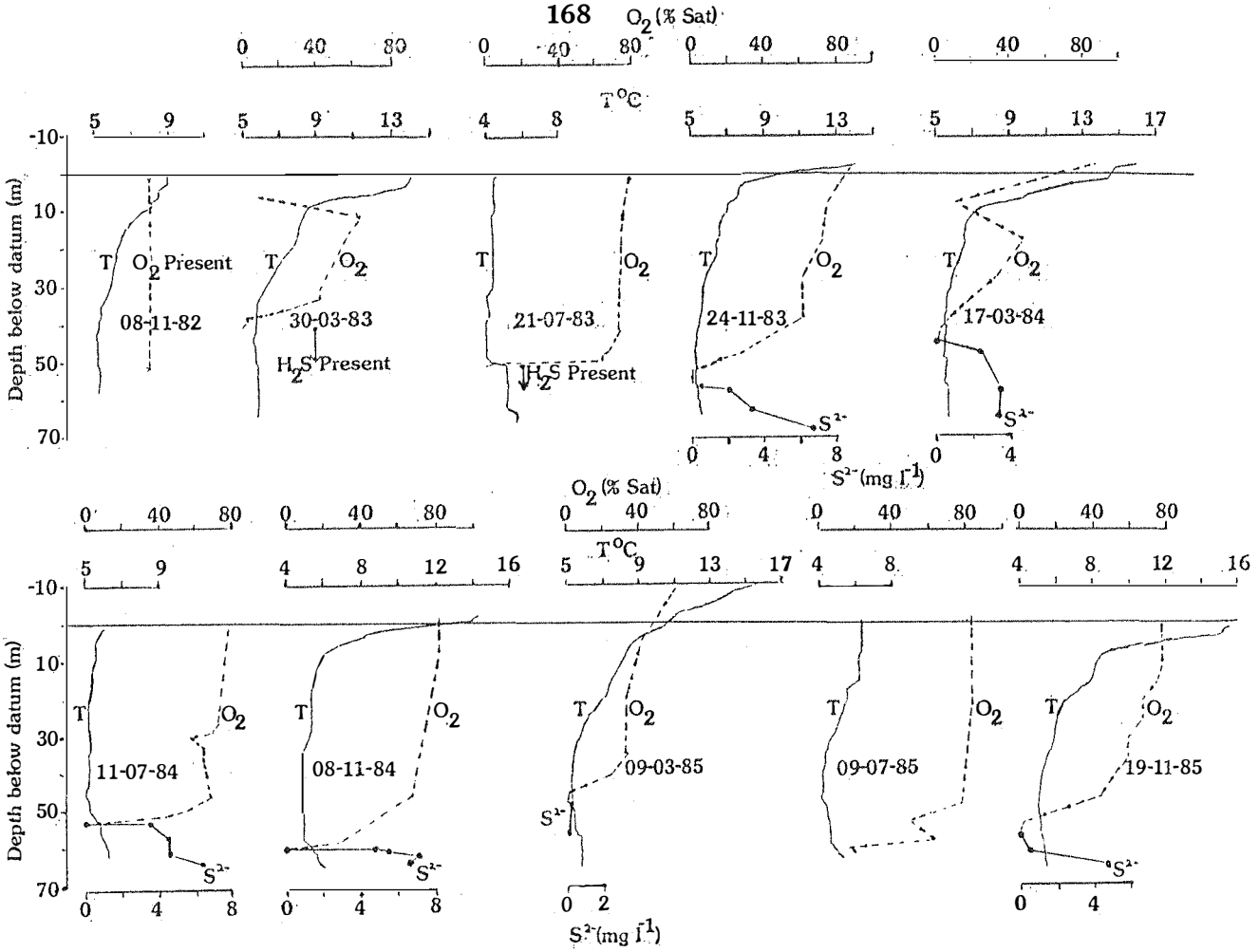


Figure 5.21 Thermal, dissolved oxygen, and dissolved sulphide profiles for Lake Murchison.

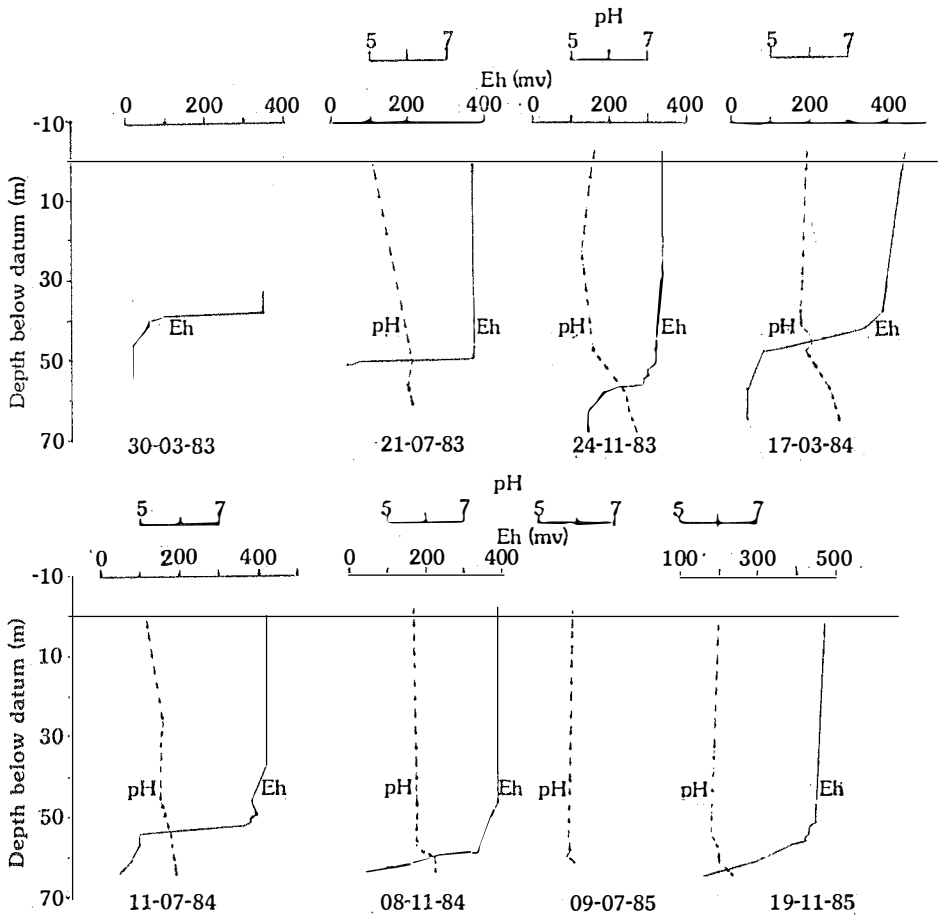


Figure 5.22 Apparent redox potential (Eh) and pH profiles for Lake Murchison.

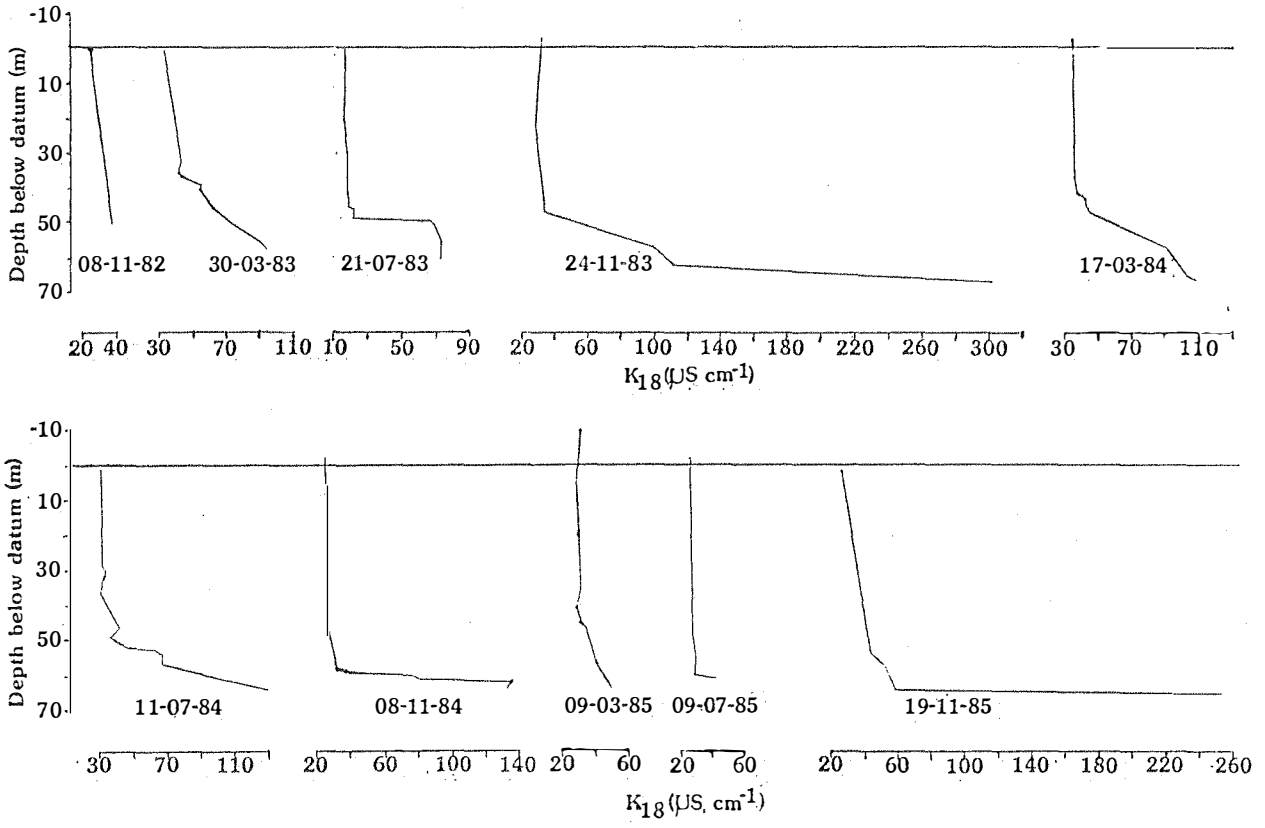


Figure 5.23 Profiles of electrical conductivity at 18°C (K_{18}) for Lake Murchison.

be attributed to sampling error. Large water level fluctuations, combined with the steep, narrow nature of the original river course, made sampling at exactly the same location every time very difficult, and not being exactly in the same position could result in an error of several metres in the maximum depth attained. This is vividly demonstrated by the profiles for 1985. In July, a maximum depth of only 61 metres datum was obtained, and chemical stratification appeared to have almost disappeared. However the following November, the maximum depth sampled was 65 metres datum, with results showing a rapid increase in K_{18} over the last two metres. The chemical stratification revealed by this increasing K_{18} was probably also present in July, but missed on that occasion as the maximum depth sampled was shallower than the stratification itself.

(e) Dissolved solutes

The dominant solutes of the monimolimnetic pool were calcium, magnesium, bicarbonate, iron, and manganese. Depth profiles for these (Figures 5.24 and 5.25) resemble those for K_{18} (Figure 5.23), with concentrations increasing rapidly near the bottom of the reservoir. The concentrations of these solutes in the monimolimnetic pool increased gradually after formation of the reservoir, reaching a maximum in November, 1983. Subsequent profiles then show decreases, apart from in November, 1985, when the maximum values for iron (51 mg L^{-1}) and manganese (7.7 mg L^{-1}) were recorded for the study. Concentrations of the other ions also increased on that occasion. However, comparisons are difficult due to the depth of measurement problems discussed in Section 5.3.5(d), above.

Potassium profiles (Figure 5.26) resemble those of calcium, magnesium, bicarbonate, iron, and manganese, but its concentrations are too low to add significantly to the chemical stratification. In comparison, concentrations of the minor anion, sulphate, fell to zero below the chemocline, due to reduction to sulphides (Figure 5.26), and sodium and chloride displayed little change in concentration throughout the depth of the reservoir (Figure 5.27).

Ternary diagrams showing ionic proportions from three occasions in 1983, reveal the chemistry of Lake Murchison changed markedly with depth. The surface waters had proportions akin to seawater, but once below the chemocline, these changed towards World Average Freshwater, as calcium and bicarbonate became the dominant ions (Figure 5.28).

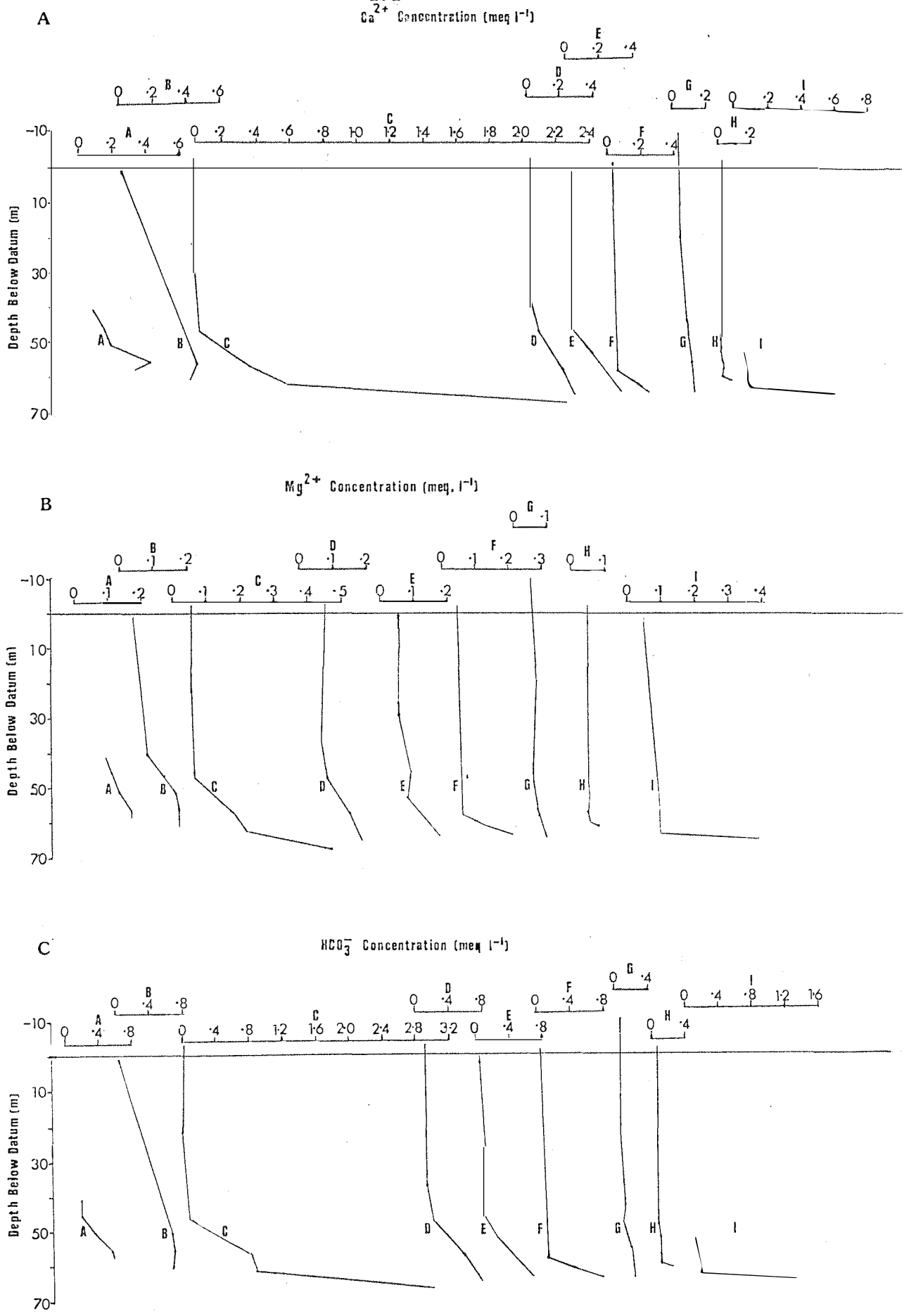


Figure 5.24 Profiles of calcium (A), magnesium (B), and bicarbonate (C) ion concentrations in Lake Murchison. A = 30-03-1983, B = 21-07-1983, C = 24-11-1983, D = 17-03-1984, E = 11-07-1984, F = 08-11-1984, G = 09-03-1985, H = 09-07-1985, I = 19-11-1985.

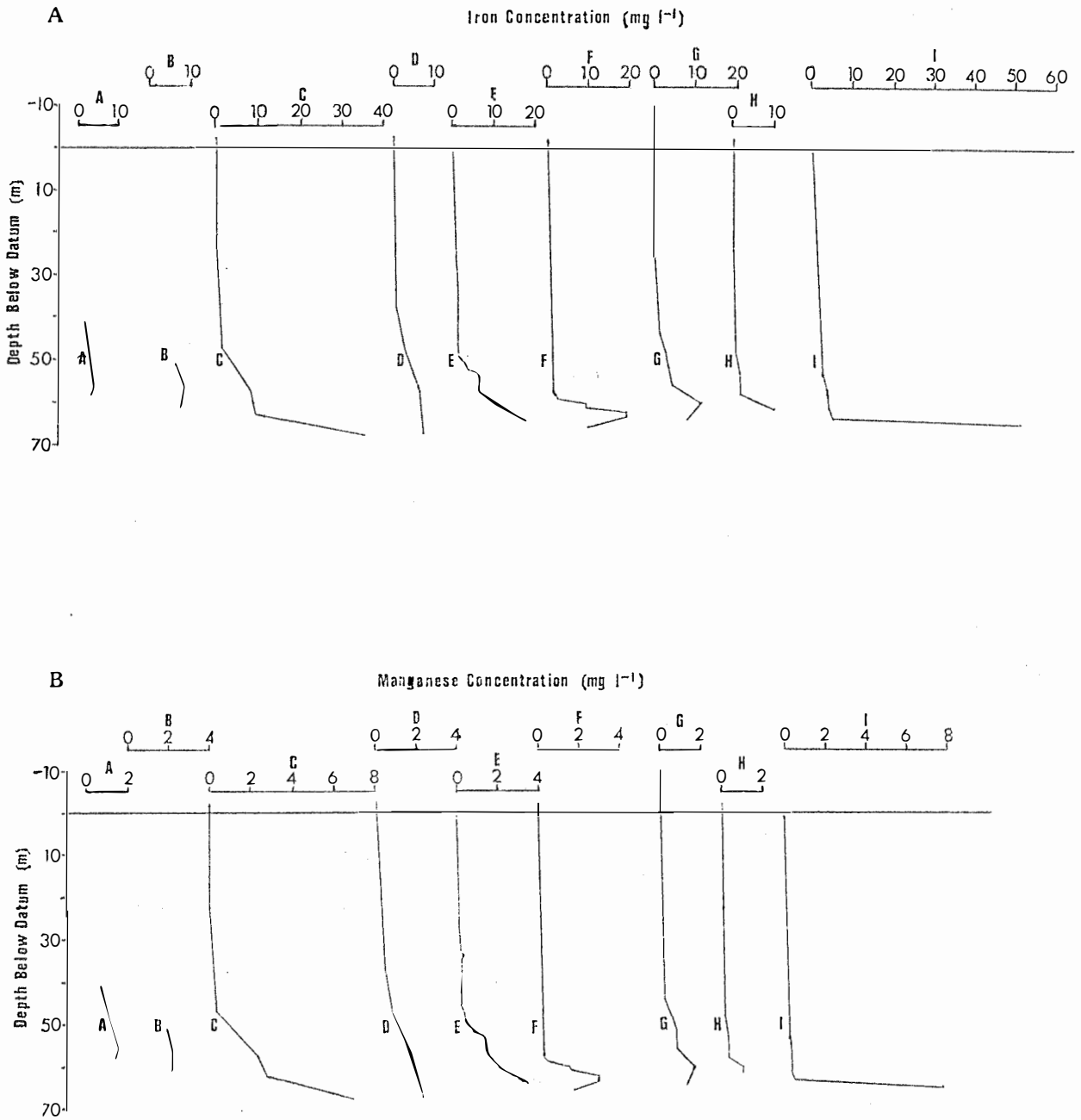


Figure 5.25 Profiles of dissolved iron (A) and manganese (B) concentrations in Lake Murchison. A = 30-03-1983, B = 21-07-1983, C = 24-11-1983, D = 17-03-1984, E = 11-07-1984, F = 08-11-1984, G = 09-03-1985, H = 09-07-1985, I = 19-11-1985.

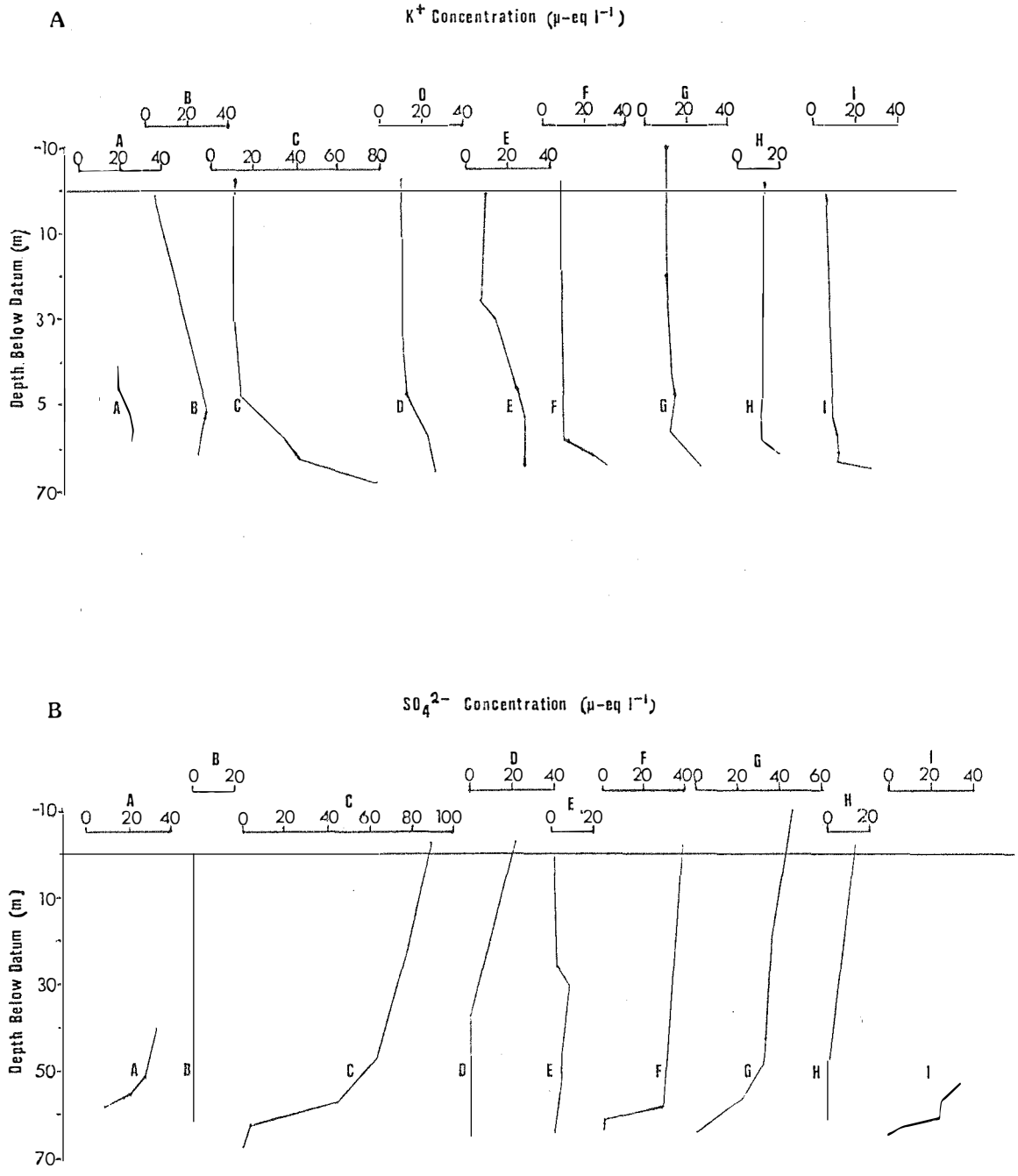


Figure 5.26 Profiles of potassium (A) and sulphate (B) ion concentrations in Lake Murchison. A = 30-03-1983, B = 21-07-1983, C = 24-11-1983, D = 17-03-1984, E = 11-07-1984, F = 08-11-1984, G = 09-03-1985, H = 09-07-1985, I = 19-11-1985.

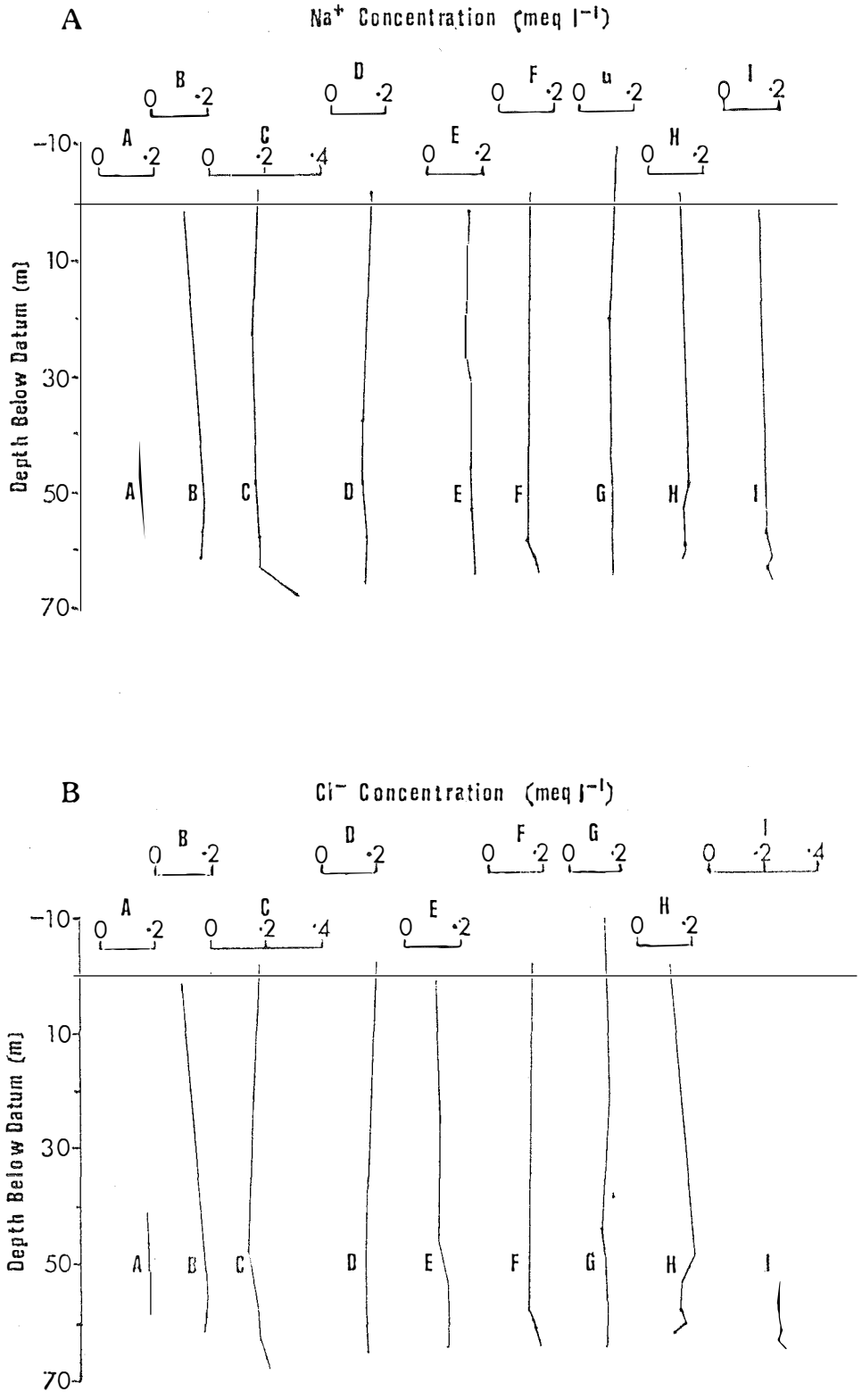


Figure 5.27 Profiles of sodium (A) and chloride (B) ion concentrations in Lake Murchison. A = 30-03-1983, B = 21-07-1983, C = 24-11-1983, D = 17-03-1984, E = 11-07-1984, F = 08-11-1984, G = 09-03-1985, H = 09-07-1985, I = 19-11-1985.

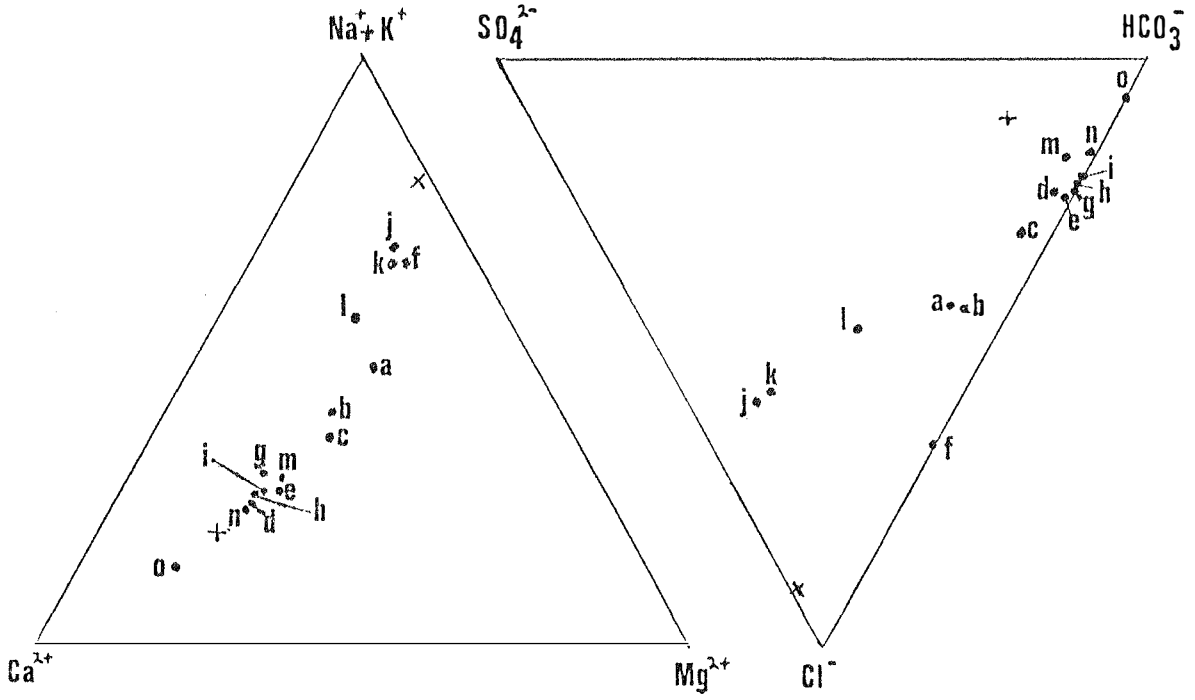


Figure 5.28 Ternary diagrams showing the ionic proportions of the major ions at various depths for the three sampling occasions in 1983.
a to e = 30-03-1983. a = 40m, b = 45m, c = 50m, d = 55m, e = 57m.
f to i = 21-07-1983. f = 0m, g = 50m, h = 55m, i = 60m.
j to o = 24-11-1983. j = 0m, k = 25m, l = 50m, m = 60m, n = 65m, o = 70m.

x = seawater and + = World Average Freshwater.

Lake Murchison remained chemically stratified throughout the three years of the study, but this gradually diminished, with the concentrations of calcium, bicarbonate, and manganese at 60 metres depth, relative to datum, steadily declining since their maximum levels in November, 1983 (Figure 5.29). Conductivity remained fairly constant at this depth for the first two years, but this too began to decline after July, 1984. The concentration of dissolved iron, on the other hand, increased slightly until early 1985, but declined after that, while sodium and chloride remained at relatively constant levels throughout the study period. The increasing concentrations of dissolved iron may have offset the falling levels of calcium and bicarbonate until July, 1984, so that conductivity remained fairly constant. The ionic composition of the waters at this depth moved progressively away from the World Average Freshwater type towards proportions similar to seawater (Figure 5.30), as the depth of the chemocline sank through this level of the reservoir. There was a slight reversal of this trend in November, 1985.

(f) The upstream extent of chemical stratification

The upstream extent of the chemical stratification in Lake Murchison was investigated in July, 1984, with measurements of water temperature and dissolved oxygen being made at points approximately 500 and 1000 metres upstream of the normal sampling point. Chemical stratification was still present 1000 metres upstream, with an increase in water temperature of 0.4°C close to the bottom (Figure 5.31). However trace amounts of oxygen (0.65% saturation) were present right to the bottom at this location, 58 metres below the surface. A thermal inversion of 1.2°C occurred close to the bottom at the point only 500 metres upstream, and oxygen was present 54 metres below the surface, but deeper waters were anoxic. A similar thermal inversion was present at the bottom close to the dam wall, but anoxia here began 52 metres below the surface.

These data reveal a wedge of colder water intruding close to the bottom of the reservoir from upstream. This cold density current, probably originating as snow-melt in the catchment and flowing down the drowned river channel, was being forced upwards and over the denser, but warmer, waters of the monimolimnetic pool, depressing its upstream end and tilting the chemocline away from the horizontal, rather than underflowing or otherwise disrupting the chemical

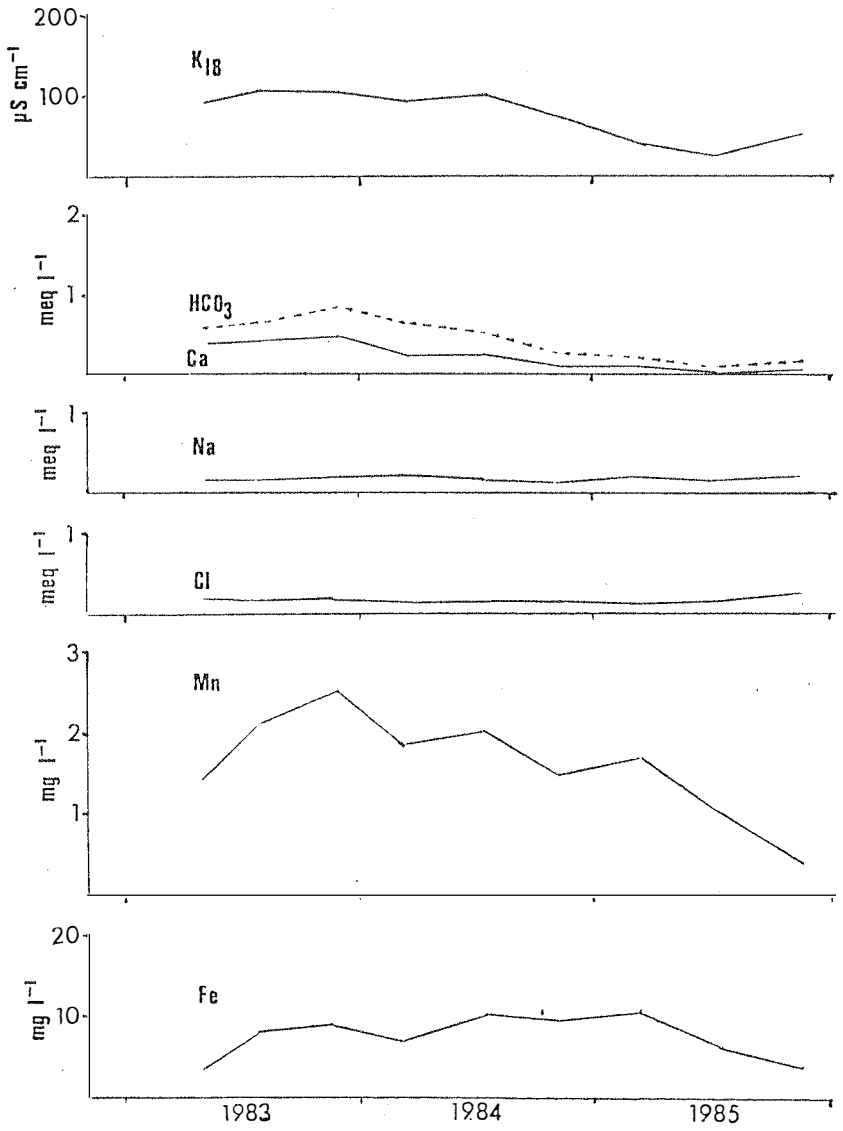


Figure 5.29 Decline in electrical conductivity at 18°C (K_{18}) and in the concentrations of various solutes at 60 metres depth, relative to datum, in Lake Murchison, over the study period.

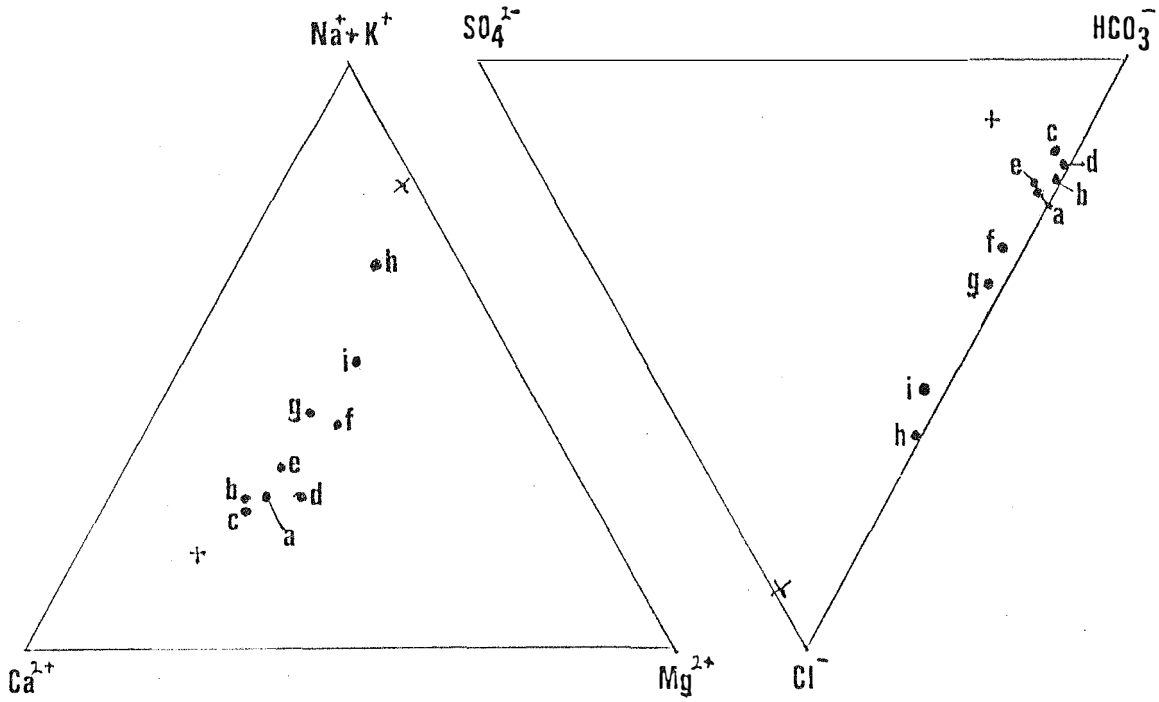


Figure 5.30 Ternary diagrams showing the temporal changes in the ionic proportions of waters from 60 metres depth, relative to datum, in Lake Murchison. Proportions have changed from close to World Average Freshwater (+) early in the study to close to seawater (X) three years later. a = 30-03-1983, b = 21-07-1983, c = 24-11-1983, d = 17-03-1984, e = 11-07-1984, f = 08-11-1984, g = 09-03-1985, h = 09-07-1985, i = 19-11-1985.

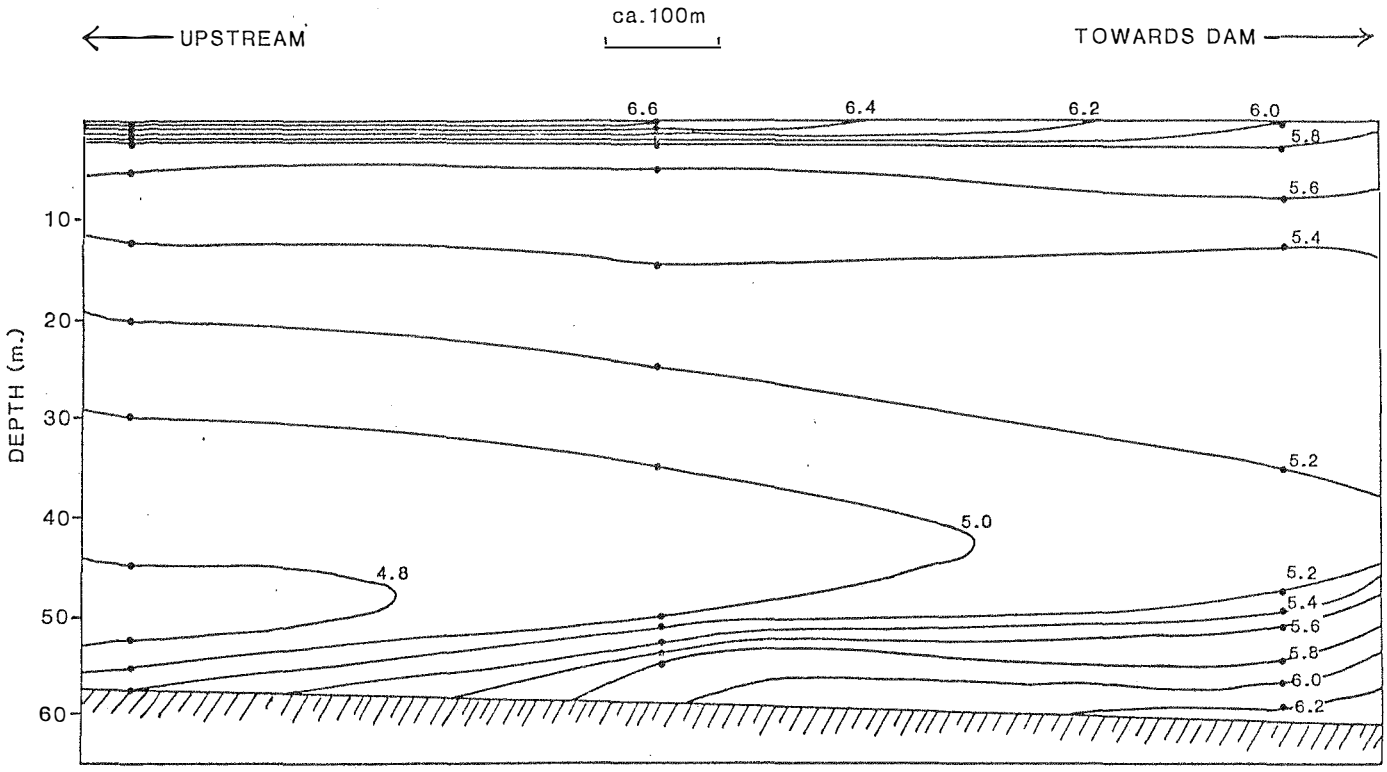


Figure 5.31 Thermal stratification in Lake Murchison over a ca. one kilometre section immediately upstream of the regular sampling location, July, 1984. An inverse thermal profile exists, indicating the presence of the monimolimnetic pool over this distance. A cold density current is overflowing the pool from upstream.

stratification. This is shown by both dichothermy and anoxia commencing at greater depths as distance upstream increases.

5.3.6 Physicochemical Features of Lake Rosebery

Lake Rosebery is also a humic reservoir, with surface g_{440} values of between 7.20 and 10.20 m^{-1}. Turbidity was always below 0.5 N.T.U. Both increased slightly near the bottom in November, 1983 and July, 1984, when interference from dissolved iron and manganese was neglibable, as the waters were oxygenated.

This reservoir also exhibited initial chemical stratification after it was formed (Figures 5.32, 5.33). It was thermally stratified in November, 1983, four months after its formation, but oxygen was present to the bottom. pH, calcium, bicarbonate, magnesium, and K_{18} all increased slightly with depth, particularly below 30 metres. Potassium and sulphate also increased with depth, but sodium and chloride remained constant. Ionic proportions changed from a seawater type order of dominance near the surface to one resembling World Average Freshwater at the bottom. Due to oxidizing conditions, dissolved iron and manganese remained below 1.0 mg L^{-1}.

Lake Rosebery was still strongly thermally stratified in March, 1984. Redox chemistry changed from oxidizing to reducing conditions ten to twenty metres below the surface, coinciding with the thermocline, and waters below this were anoxic and sulpheretted. Dissolved iron and manganese concentrations increased with depth from below the thermocline, but pH, K_{18}, calcium, magnesium, and bicarbonate only showed rapid increases below 40 metres. Thus the major stratification of solutes began at this depth rather than from immediately below the thermocline and redoxcline. Again, potassium increased slightly with depth, while sodium and chloride remained at fairly constant levels; however sulphate concentrations fell to zero below 40 metres. Ionic proportions also changed from a seawater type order of dominance to a World Average Freshwater type below 40 metres.

This incipient monimolimnetic pool had vanished by July, 1984, when the lake was isothermal with almost 50% saturation dissolved oxygen near the bottom. Major ions changed little with depth, and sodium and chloride were always dominant.

5.3.7 Physicochemical Features of Lake Pieman

Limnological investigations of this fourth humic reservoir commenced in mid September, 1986. Thermal statification was just

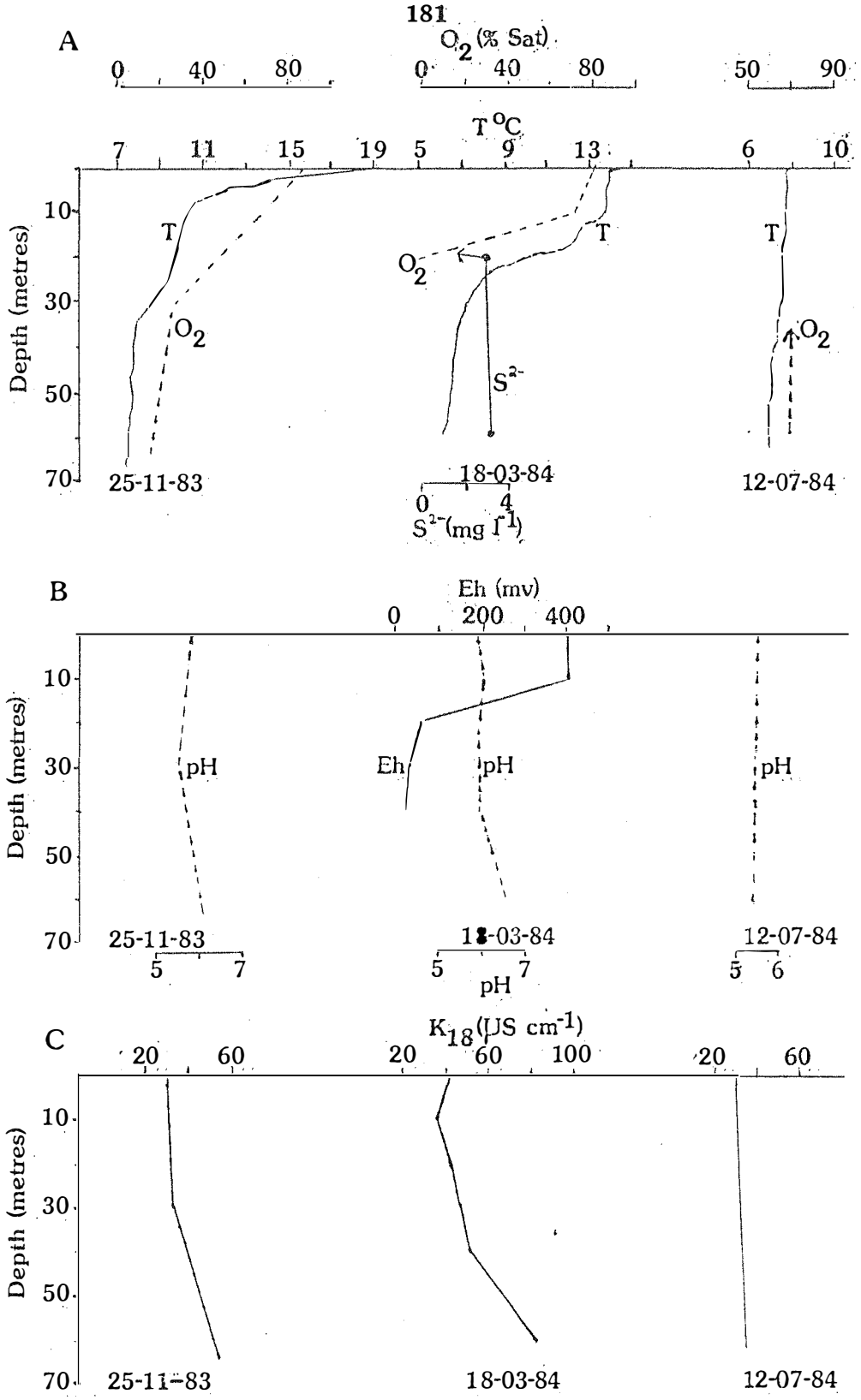


Figure 5.32 Thermal, dissolved oxygen, and dissolved sulphide profiles (A); pH and apparent redox potential (Eh) profiles (B); and electrical conductivity at 18°C (K_{18}) profiles (C), for Lake Rosebery.

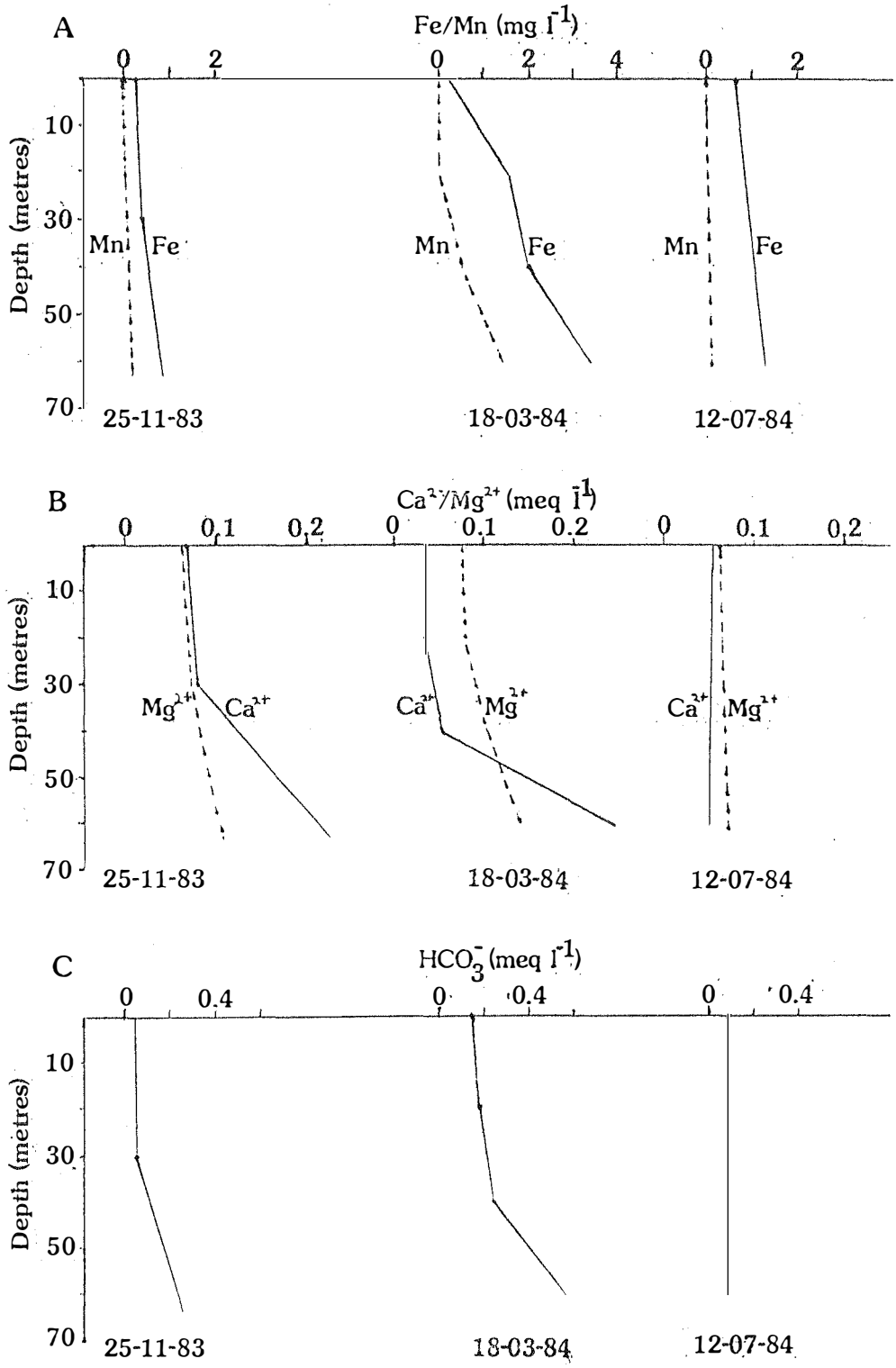


Figure 5.33 Dissolved iron and manganese profiles (A); calcium and magnesium ion profiles (B); and bicarbonate ion profiles (C) for Lake Rosebery.

starting, and probably the reservoir had been circulating since impoundment. However, temperatures increased 0.1°C below 80 metres, while dissolved oxygen diminished from 84% saturation at the surface to 27% at 94 metres, and K_{18} increased from 49 to 64 $\mu S\ cm^{-1}$ (Figure 5.34). Concentrations of calcium, magnesium, sulphate, bicarbonate, iron, and manganese also increased slightly between these depths, indicating the onset of chemical stratification.

Measurements at the height of thermal stratification, at the end of January, 1987, confirmed the development of incipient meromixis in Lake Pieman (Figure 5.34, 5.35). The reservoir was anoxic and reducing, with considerable amounts of dissolved sulphides present, below 60 metres. Conductivity and dissolved iron and manganese also began increasing below this depth, although rises in temperature and concentrations of magnesium, calcium, and bicarbonate did not occur until deeper, in a manner similar to Lake Rosebery. However, this chemical stratification did not persist. Measurements in September, 1987, showed no trace of a dichothermal temperature profile or conductivity differences with depth, and oxygen was present right to the bottom.

5.3.8 Physicochemical Features of Lake Barrington

Lake Barrington is a moderately humic reservoir, with surface gilvin levels between 2.00 and 3.60 m^{-1}. Colour increases slightly in the mid-depths, but decreases below the chemocline to as low as 1.50 m^{-1}. Surface turbidities are low (1.2 N.T.U.), but measurements of turbidity in the bottom waters were impossible due to interference by precipitates of iron and manganese.

This reservoir was sampled annually between 1982 and 1985, and the results are shown in Figures 5.36, 5.37, and 5,38. The reservoir displayed strong localized meromixis throughout the study period, with a monimolimnetic pool three to four metres deep.

Thermal profiles (Figure 5.36a) were markedly dichothermal, with temperature increases of over 1.0°C close to the reservoir bottom. Concomitant with this, oxygen concentrations fell from greater than 60% saturation to zero in less than one metre, and up to 13 mg L^{-1} of dissolved sulphides were present within the monimolimnetic pool. Very rapid changes in redox potential (Eh) also took place, from oxidizing to reducing conditions, and pH increased slightly.

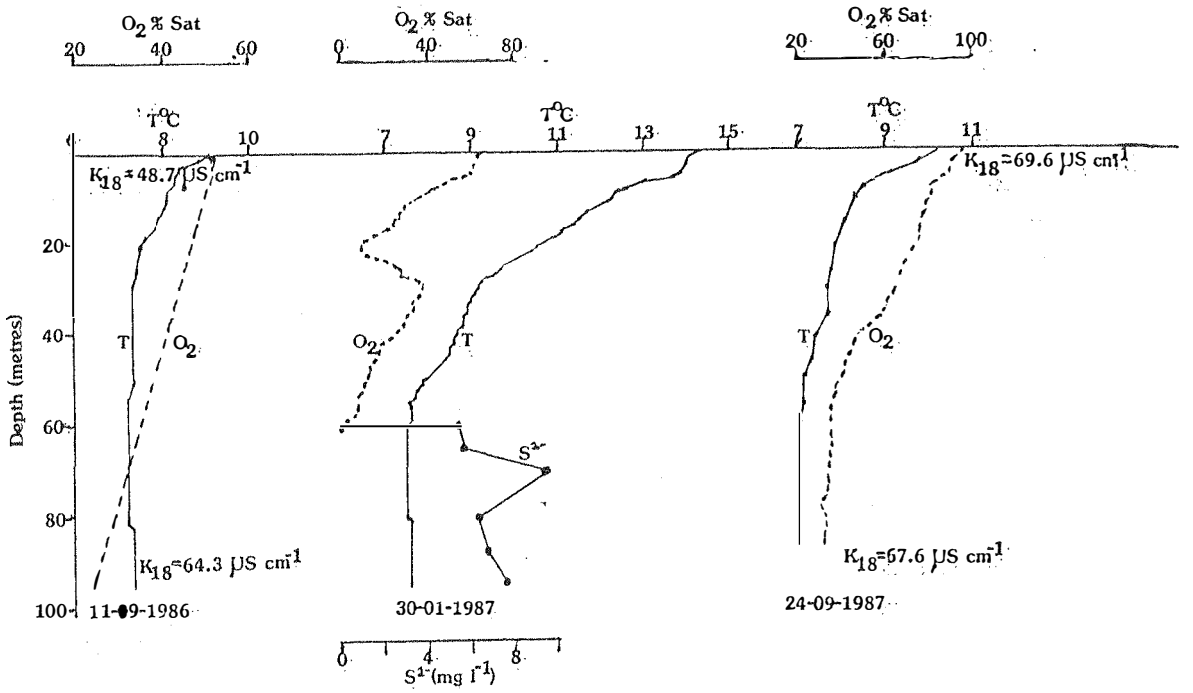


Figure 5.34 Thermal, dissolved oxygen, and dissolved sulphide profiles for Lake Pieman. K_{18} is also indicated on two occasions.

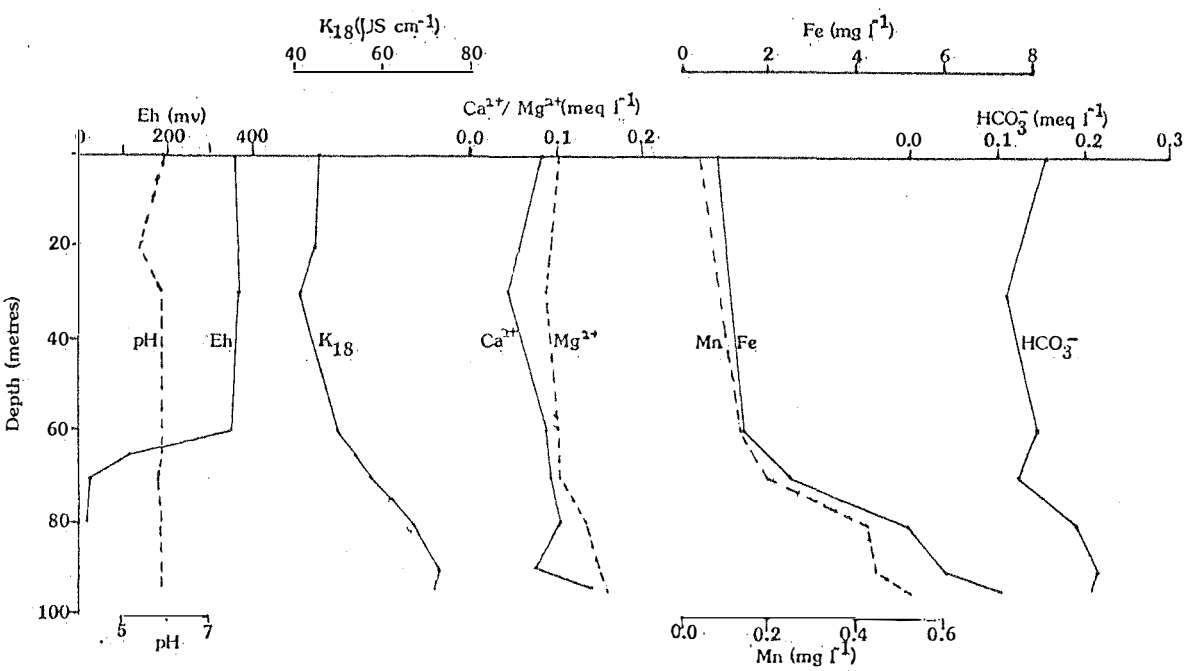


Figure 5.35 Depth profiles of pH; apparent redox potential (Eh); electrical conductivity (K_{18}); calcium, magnesium, and bicarbonate ion concentrations; and dissolved iron and manganese concentrations in Lake Pieman for 30-01-1987.

The presence of chemical stratification in Lake Barrington is highlighted by rises in electrical conductivity (K_{18}), and by the concentrations of some major ions and dissolved iron and manganese. The K_{18} increase is extremely sharp and rapid, occurring over a stratum only one metre deep (Figure 5.37a), and then increasing to almost 500 μS cm^{-1} at the bottom. Identical patterns are shown by dissolved iron and manganese, which increase from almost zero to 80 mg L^{-1} and 10 mg L^{-1} respectively (Figure 5.37b,c). In September, 1984, dissolved iron exceeded 120 mg L^{-1} and manganese 12 mg L^{-1} at the deepest point sampled.

Figure 5.38 gives depth profiles for calcium, magnesium, and bicarbonate. Lake Barrington differs from the reservoirs of the Pieman scheme in that magnesium rather than calcium is the dominant cation. Concentrations of these two ions also increased rapidly within the monimolimnetic pool, as did bicarbonate. These ions, along with dissolved iron and manganese, were the major solutes causing the chemical stratification. Levels of sodium, potassium, and chloride increased only slightly, while sulphate values fell to zero below the chemocline.

5.3.9 Volume Weighed Average Temperature and Oxygen; Heat Contents, Thermal Stabilities and Birgean Wind Work in the Five Reservoirs

The results of these parameters, calculated for all five reservoirs, are given in Table 5.5. The volume weighed average temperatures show the impoundments, and especially Lake Murchison, are quite cold. This is so even in summer, after considerable amounts of heat have been absorbed, mainly by the humic surface waters, because the hypolimnion remains cold and this lowers the volume weighed mean temperature. Although sampling was too infrequent to allow accurate calculations of the annual heat budgets, θ_{ba}, some estimates of the amount of heat gained or lost by some of the reservoirs are still possible. For Lake Murchison this was at least 10085 cals cm^{-2} for the 1983-84 heating period, and 10891 cals cm^{-2} for the 1984-85 period, while that for Lake Rosebery, estimated from heat lost during autumn, 1984, was 10635 cals cm^{-2}. The variability in the heat contents of the other reservoirs indicate considerable heat exchange by their waters, too. Volume weighed average oxygen contents showed the reverse trend to that for temperature, being highest during winter measurements and lowest in summer due to hypolimnetic oxygen demand.

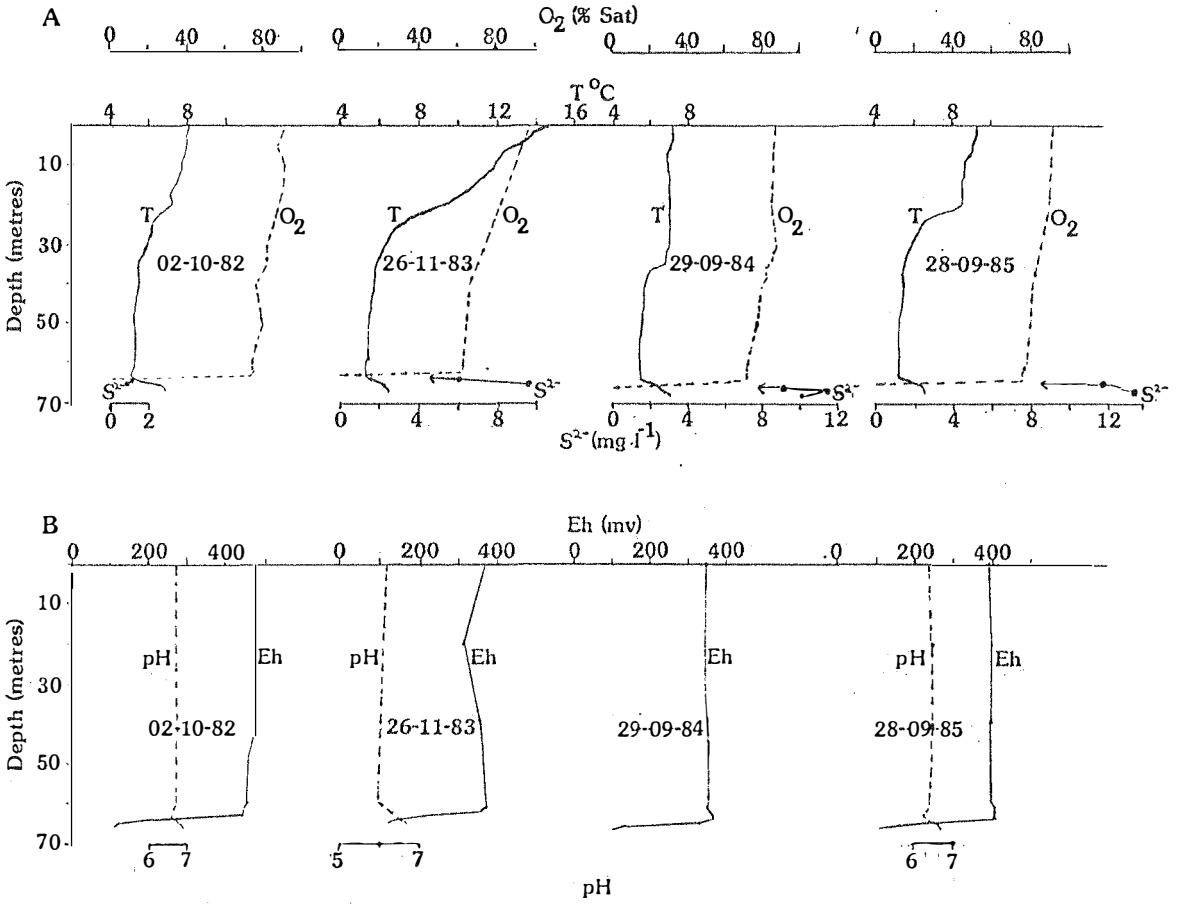


Figure 5.36 Thermal, dissolved oxygen, and dissolved sulphide profiles (A); and of pH and apparent redox potential (Eh) (B) in Lake Barrington.

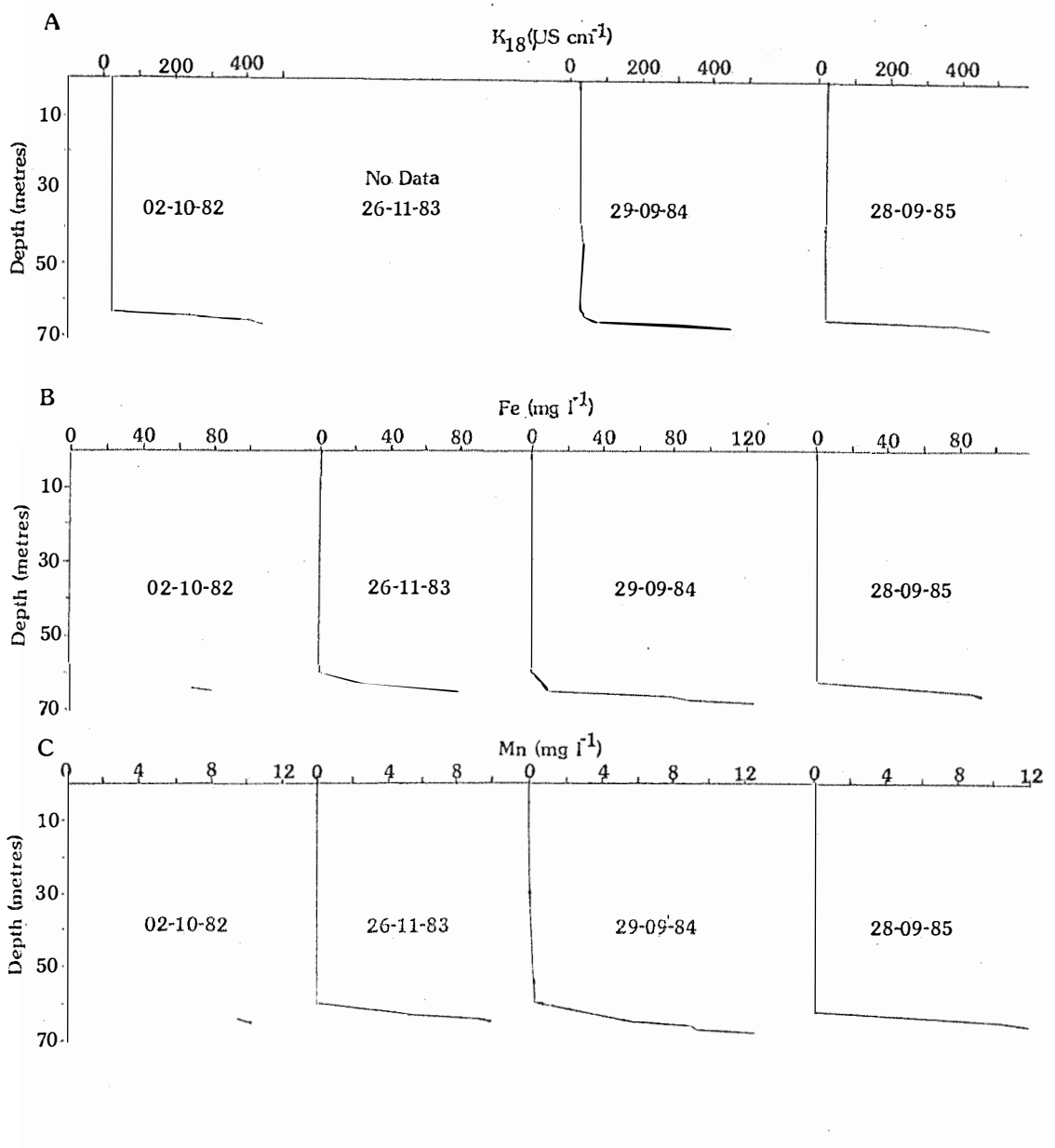


Figure 5.37 Profiles of electrical conductivity (K_{18}) (A), dissolved iron (B), and dissolved manganese (C) in Lake Barrington.

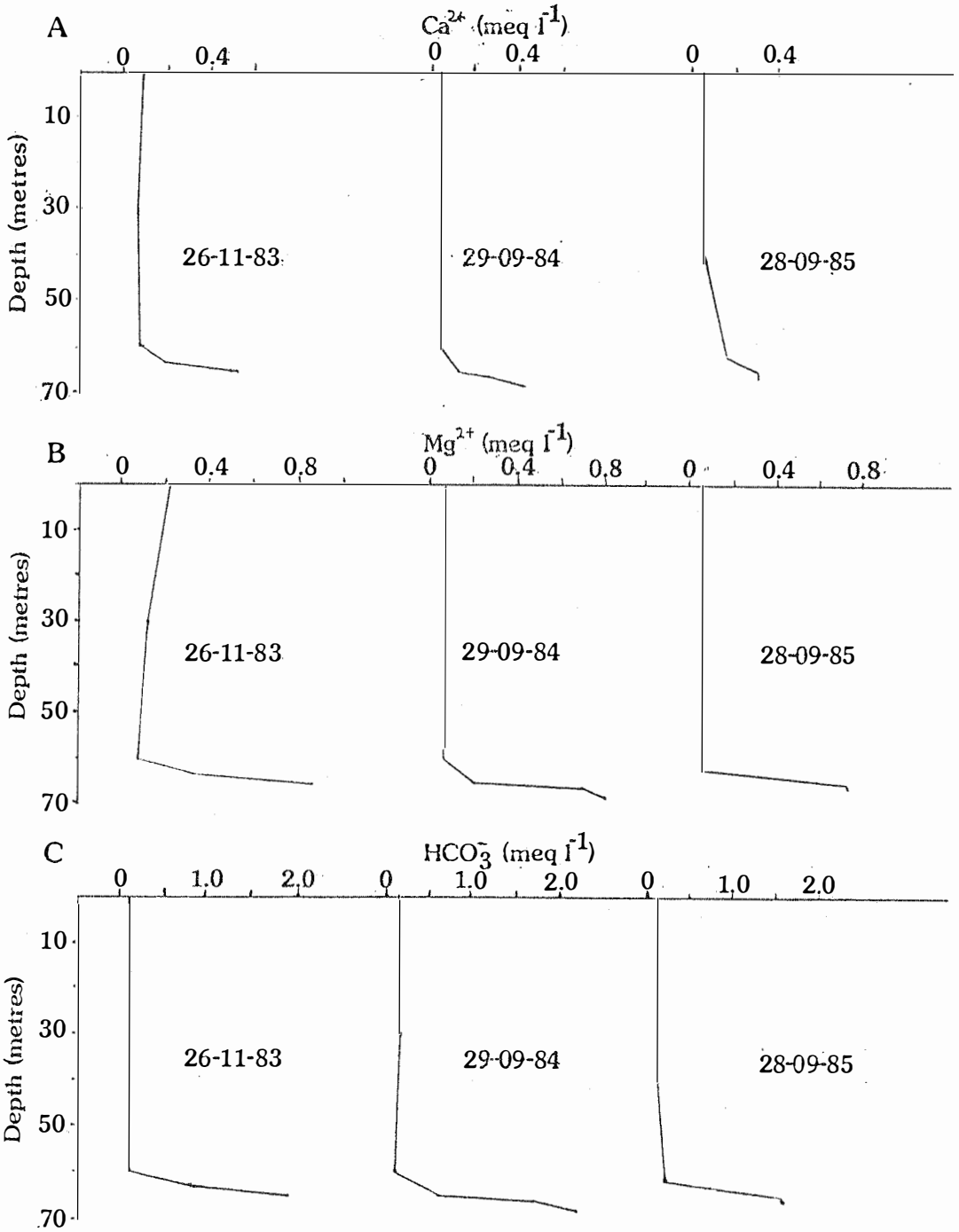


Figure 5.38 Calcium (A), magnesium (B), and bicarbonate (C) ion concentration profiles for Lake Barrington.

Table 5.5: Volume weighted mean temperatures and oxygen contents, heat contents, thermal stabilities, and Birgean Wind Work values for Lake Barrington and the Pieman River reservoirs, N.S. = Not Stratified

Lake	Date	Average Oxygen	Average Temp.	Heat Content	Thermal Stability	Birgean Wind Work
		mg L^{-1}	°C	Cals cm^{-2}	gm-cm cm^{-2}	gm-cm cm^{-2}
Barrington	02-10-82	10.48	7.0	18919	139	233
	26-11-83	9.20	9.9	26603	791	751
	29-09-84	10.39	6.9	18964	N.S.	N.S.
	28-09-85	10.39	7.6	21866	261	366
Murchison	08-11-82	-	7.1	14633	145	149
	30-03-83	4.77	8.8	17936	450	289
	21-07-83	9.59	4.3	8831	N.S.	N.S.
	24-11-83	8.31	7.5	15818	280	163
	17-03-84	4.04	8.9	18916	660	293
	11-07-84	9.11	5.5	11143	N.S.	N.S.
	08-11-84	9.29	7.2	15130	378	106
	09-03-85	4.78	9.6	22034	738	505
	09-07-85	10.22	5.8	12137	44	58
	19-11-85	8.11	8.7	17586	546	238
Rosebery	25-11-83	5.79	11.0	18965	401	567
	18-03-84	4.67	14.1	23382	558	1186
	12-07-84	-	7.7	12747	13.5	221
Mackintosh	11-03-81	1.25	12.1	28496	649	1179
	10-06-81	4.36	10.0	20238	83	614
	20-08-81	4.37	7.9	19036	50	379
	20-04-82	5.02	12.8	23369	257	928
	07-11-82	-	8.7	16933	154	275
	24-11-83	2.99	11.1	16840	191	373
Pieman	11-09-86	8.12	7.8	22403	137	621
	30-01-87	3.36	10.9	31140	1077	1495
	24-09-87	-	8.2	21796	222	592

All the reservoirs develop quite high summer thermal stabilities, and large inputs of energy would be required at these times to completely homogenize them. Lake Mackintosh is possibly the least stable of them. Thermal stabilities are low during winter circulation, and could not be calculated for two occasions for Lake Murchison and once for Lake Barrington, because of insufficient density differences.

Birgean Wind Work calculations follow those for stability. However values for Lakes Mackintosh, Rosebery and Pieman were generally higher than those for Lake Murchison, and possibly Lake Barrington. This indicates more energy is expended to produce the thermal stratification patterns observed within these three reservoirs, than in the other two.

PART C THE LAKES OF THE LOWER GORDON RIVER AREA

5.3.10 Physicochemical Features of Lake Fidler

Thermal and chemical profiles for Lake Fidler from 1983 onwards are given in Figure 5.39. All depths are relative to the gaugeboard datum for this lake. Thermal stratification is evident in the mixolimnion of the lake during summer. A slight mesothermal bulge, of less than 1°C, was present centred 3.5 metres deep in April, 1983, and at 3.0 metres in both April 1986 and April 1987. However, the depth of the oxycline varied from 1985 onwards. The lake became anoxic 3.30, 3.45, and 3.55 metres deep in January and March, 1985, and April, 1986, respectively, but the maximum depth of the oxic zone was shallower, at 3.0 metres, during early 1987.

The considerable discontinuity in the pH profile in January 1985 (Figure 5.39) is indicative of the abruptness of the chemocline on that occasion, but usually pH changes were less dramatic, and so were not good indicators of chemocline position. The depth of major redox changes from oxidizing to reducing values, increased from about 3.5 metres in early 1983, to below 3.75 metres in 1986. However in February and April, 1987, the redoxcline was again positioned at a depth of 3.0 metres. The progressive deepening of the redoxcline since 1981, and its dramatic elevation again by 1987, is shown in Figure 5.40.

Profiles of conductivity (Figure 5.39) show little evidence of increased depth in the conductivity cline. However two significant

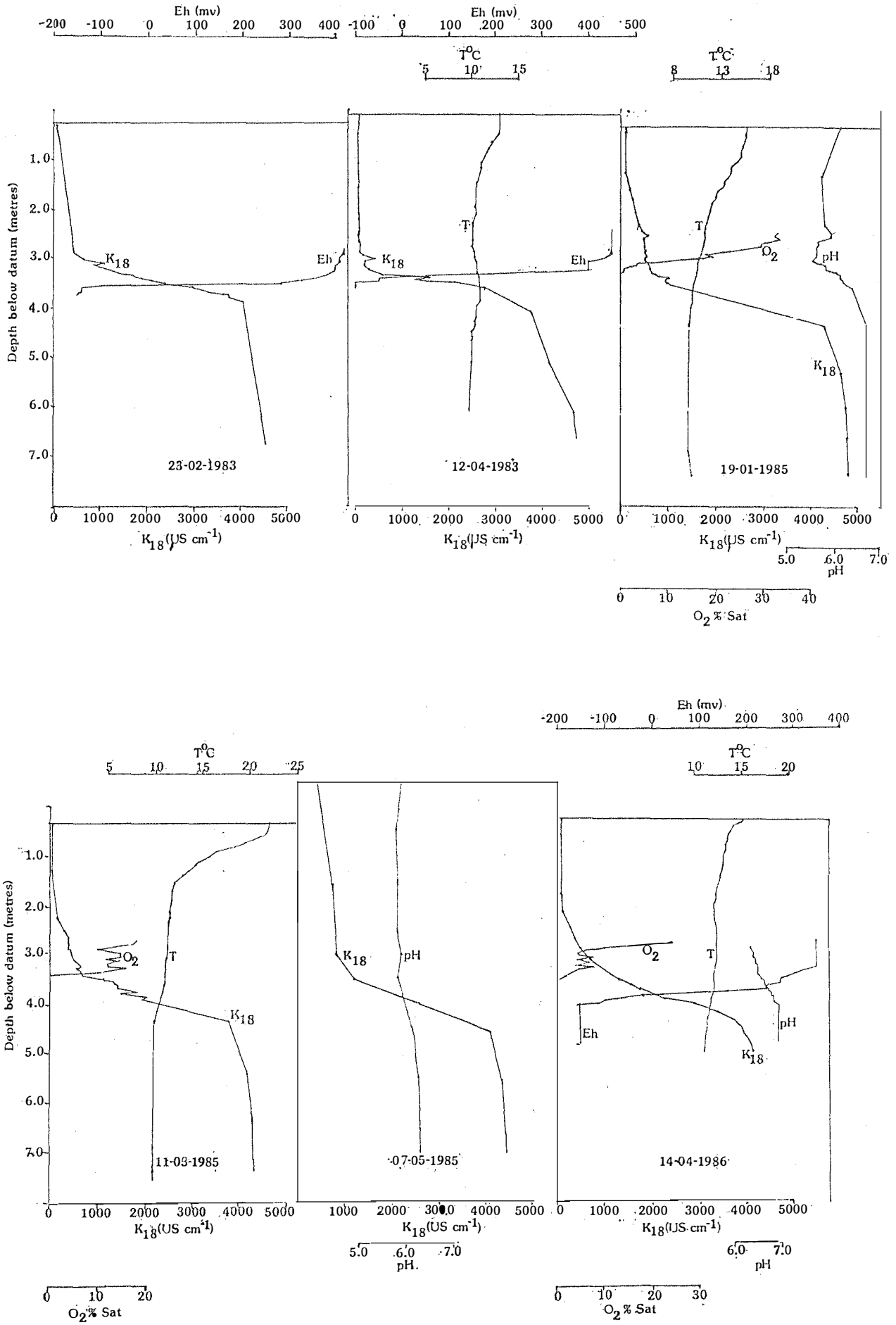


Figure 5.39 Depth profiles of various physico-chemical parameters from Lake Fidler, plotted relative to datum to show fluctuations in water level (The figure is continued next page).

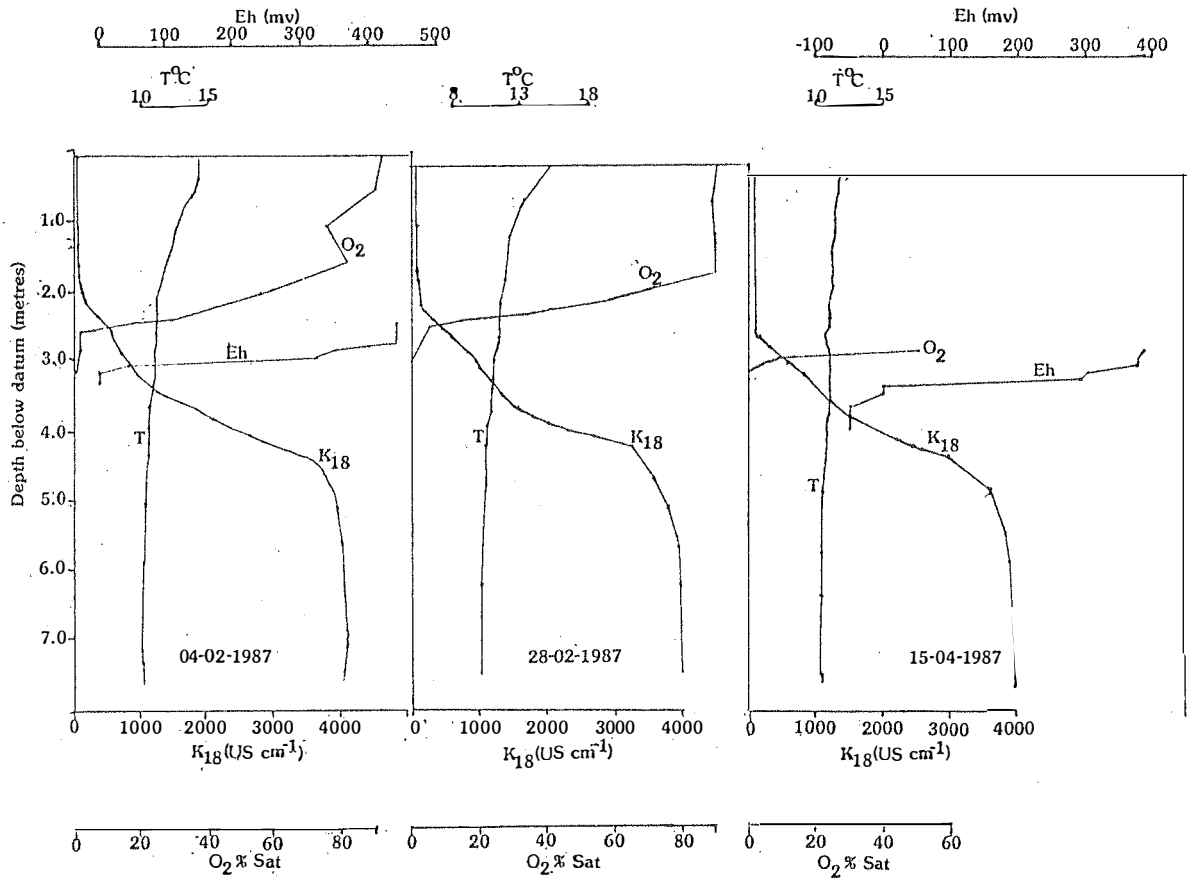


Figure 5.39 (continued). Further depth profiles of various physico-chemical parameters from Lake Fidler, plotted relative to datum.

changes are apparent. Firstly, the conductivity gradually thickened, and spanned an approximately 2.75 metre stratum in 1987, compared to a 1.50 metre stratum in January, 1985. The conductivity was also 0.75 metres shallower in 1987, and conductivity at 3.00 metres had increased by about 300-400 μS cm^{-1} (Figure 5.40). This change in the thickness and depth of conductivity indicates the possible intrusion of an interflow more saline and denser than the mixolimnetic waters, but less saline and dense than the monimolimnetic waters, positioning itself in the vicinity of the chemocline. However in May, 1985, mixolimnetic conductivities generally exceeded 500 μS cm^{-1}, and were considerably higher than those of the previous March. Changes in conductivity also occupied a narrower stratum typical of earlier measurements. This suggests an inflow of higher salinity water into the mixolimnion, elevating its conductivity on this occasion.

The second significant change is the decline of conductivities (and therefore salinities) in the bottom waters of Lake Fidler, from 4780 μS cm^{-1} at 7.0 metres in January, 1985, to 3940 μS cm^{-1} in April 1987 (Figure 5.39). This dilution must result from water movements across the chemocline, perhaps in response to the salinity gradient, and may also cause the increased thickness of the conductivity cline.

5.3.11 Physicochemical Features of Sulphide Pool

Physicochemical profiles for Sulphide Pool are shown in Figure 5.41. All depths reported below are relative to the gaugeboard datum for this lake, too. Considerable changes in its meromictic condition are apparent from December, 1981, onwards. At that time, and in February, 1982, the lake was thermally stratified, and the redoxcline was less than 2.0 metres deep. This stratum, occupying the mid depths of the lake, was also a zone of considerable pH and conductivity change. The K_{18} of the monimolimnetic waters exceeded 1600 μS cm^{-1}.

Considerable instability of the water column was apparent in April, 1982. Eh fell suddenly to near reducing conditions at 1.95 metres, returned rapidly to oxidized values at 2.00 metres, before changing once again to reducing conditions at 2.10 metres. These changes were accompanied by irregularities in the conductivity profile and decreasing pH. This pattern was confirmed by another profile, taken with a second close-interval sampler, at a different position

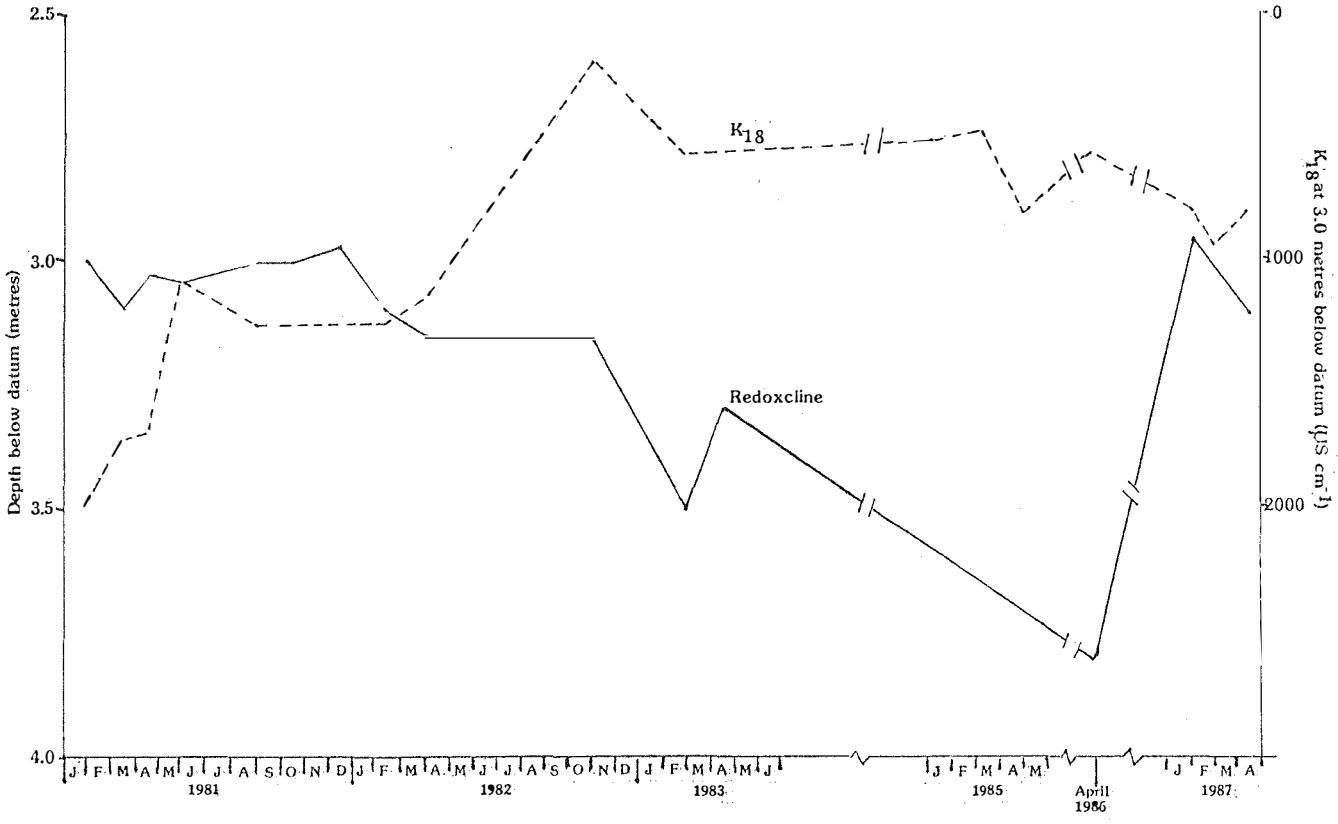


Figure 5.40 The variation in the depth, relative to datum, of the major change in Eh (redoxcline), and in the electrical conductivity (K_{18}) at a depth of 3.0 metres (relative to datum) in Lake Fidler. Data from 1981 from Bowling (1981); and for 1982 from Croome (1984).

in the lake. A mesothermal bulge was also present between 1.75 and 2.40 metres. These data indicate a considerable drop in the chemocline position took place between February and April, 1982. The chemocline was even deeper in November, 1982, and considerable instability was again apparent in the Eh and K_{18} profiles, this time within the monimolimnion.

Marked irregularities were also present in the conductivity profiles of both February and April, 1983, although the major shift in Eh remained at a fairly constant depth during this time. The chemocline may have been protected from errosive mixing by the thermal stratification present over summer. However, the concentrations of monimolimnetic solutes continually declined, and by August, 1983, the chemocline had sunk to about 2.40 metres. A slight temperature inversion and a sharp increase in K_{18} took place at this depth on this occasion.

The next data, from January, 1985, show considerable changes had taken place in Sulphide Pool, with the K_{18} at most depths being considerably higher than in August, 1983. The gradual increase in K_{18} from surface to bottom, rather than a sharp increase at depth, and the persistance of oxygen at trace amounts almost to the bottom is indicative of an inflow of slightly saline Gordon River water, not much prior to the sampling time. By March, 1985, the conductivity cline had been eroded considerably, and greater amounts of oxygen were present, especially in the mid-depths of the lake, due to mixing in the surface waters. A further saline inflow had occurred by May 1985, again raising the conductivity of the whole water column, especially below 0.5 metres. The increased K_{18} below this depth was accompanied by increases in pH, decreases in g_{440}, and increases in oxygen, to levels greater than 80% saturation close to the bottom. The lake was isothermal.

Sulphide Pool was again sampled in April 1986, and in February 1987, and was very similar on both occasions. No oxygen was present below about 2.00 metres, and conductivity increased steadily from this depth to the bottom. K_{18} data for 1986 shows two step-like increases with depth, indicating two separate strata of differing salinity water. The conductivity of the bottom waters had declined by February 1987.

Changes in the meromixis of Sulphide Pool are summarised in Figure 5.42. This illustrates a sinking redoxcline between 1981 and

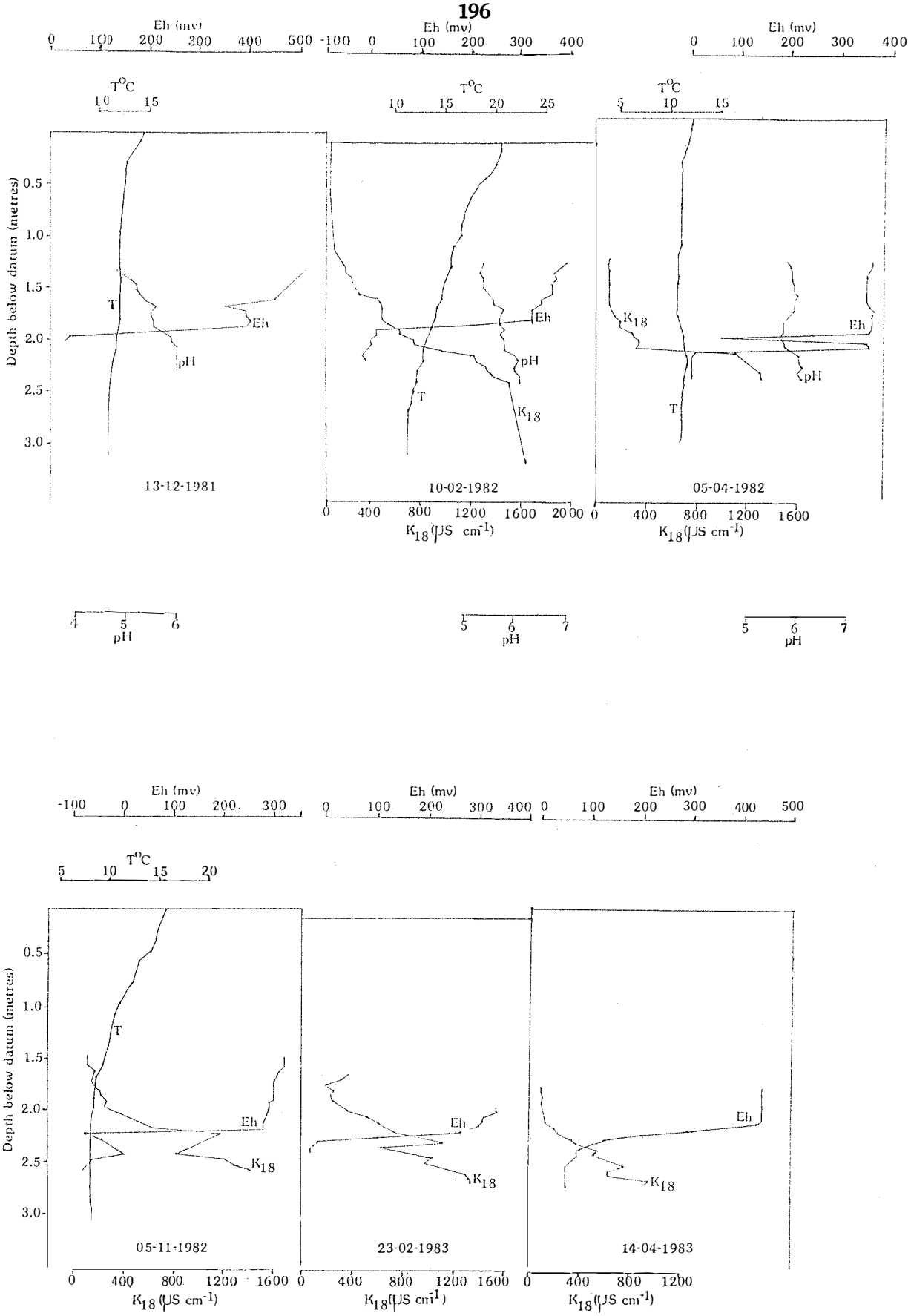
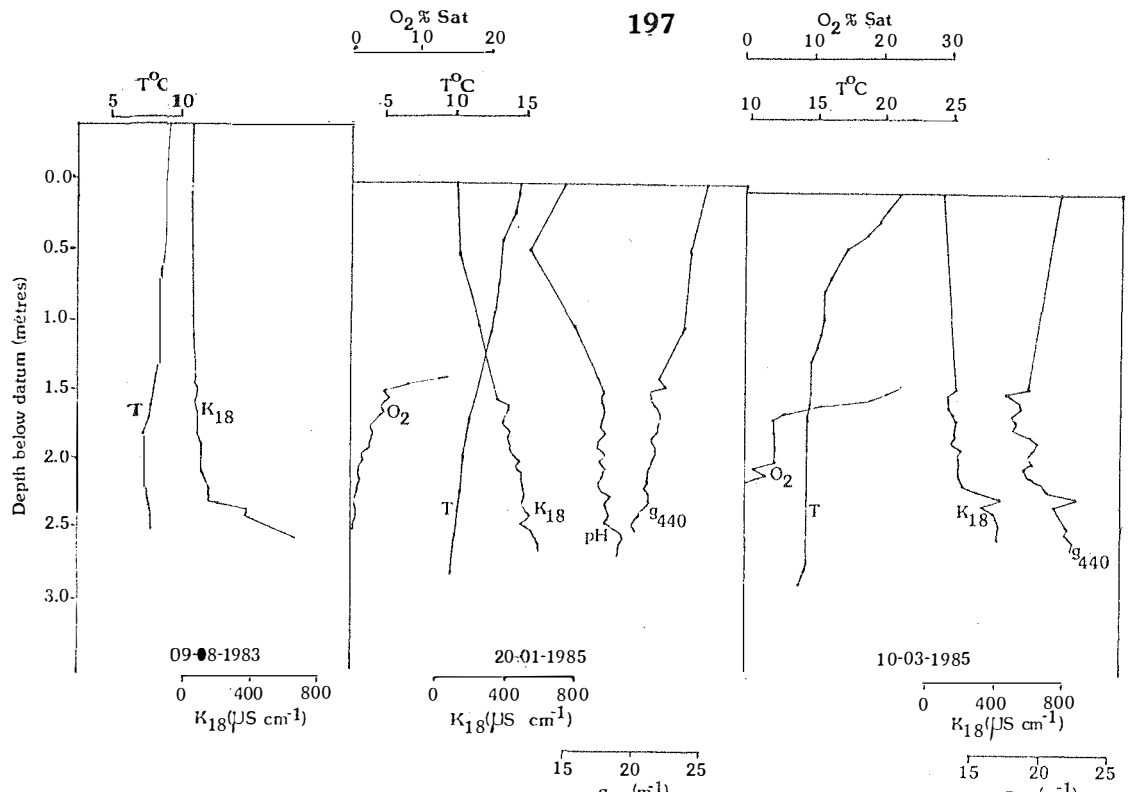


Figure 5.41 Depth profiles of various physico-chemical parameters from Sulphide Pool, plotted relative to datum to show fluctuations in water level. (The figure is continued next page.)



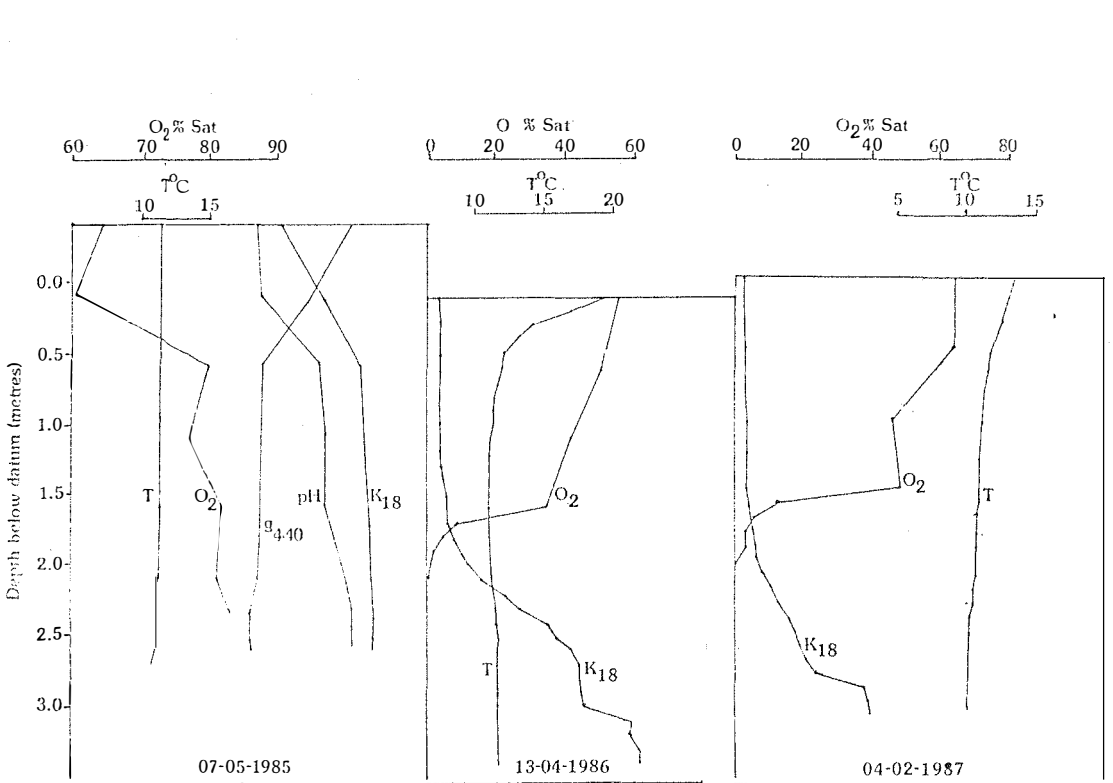


Figure 5.41 (continued) Further depth profiles of various physico-chemical parameters from Sulphide Pool, plotted relative to datum.

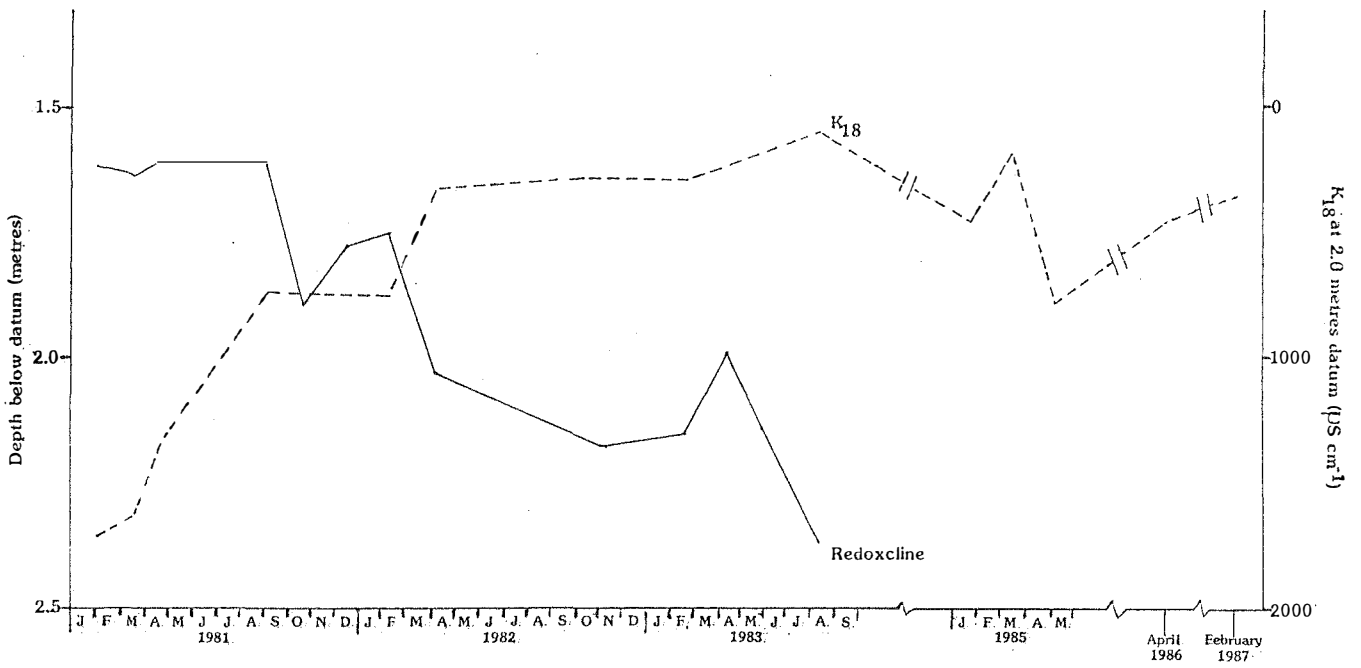


Figure 5.42 The variation in the depth, relative to datum, of the major shift in Eh (redoxcline), and in the electrical conductivity (K_{18}) at a depth of 2.0 metres (relative to datum) in Sulphide Pool. No further redox data was available after August, 1983. Data from 1981 from Bowling (1981).

1983 (no redox data were available after that date), while the conductivity at 2.00 metres also decreased as the chemocline sank through this depth. Data since 1983 show K_{18} varied at this depth due to saline inflows, but despite these, it never reached the values first recorded in early 1981.

5.3.12 Physicochemical Features of Lake Morrison

The meromictic condition of Lake Morrison decayed following the commissioning of a hydro-electric power station on the mid-reaches of the Gordon River, and its evenutal holomixis occurred in April 1978 (King, 1980; King and Tyler, 1982b, 1983). The lake has since vacillated between holomixis, when it is completely isohaline, and incipient meromixis, when conductivity clines are present (Bowling, 1981). Subsequent data from the lake are shown in Figure 5.43. It was chemically stratified from the end of 1981 to mid 1982, as evidenced by the anoxic, sulpheretted bottom waters of December, 1981, and the weak conductivity gradient present in April, 1982. However, the lake was holomictic again in November, 1982.

The three samplings in 1985 and 1986 all showed chemical stratification. Conductivity increased gradually from surface to bottom in March, 1985, while gilvin decreased with depth, and dissolved oxygen persisted almost to the sediments. The conductivity cline was similar in May, 1985, but values were higher than two months earlier. g_{440} again declined slightly with depth, while oxygen was at a minimum 1.50 metres deep, but increased markedly to almost 65% saturation near the bottom. These data suggest a large inflow of saline water, with conductivities up to 1645 μS cm^{-1}, causing chemical stratification, sometime prior to the sampling occasion. The incipient meromixis of April 1986 was accompanied on this occasion by anoxia and hydrogen sulphide, and a slight increase in temperature, in the bottom waters.

5.3.13 Calculations of Meromictic Stability

Meromictic stabilities in Lake Fidler have fallen gradually since 1977 (Figure 5.44). Initially this was fairly rapid, decreasing from 92.01 gm-cm cm^{-2} in January 1977, to 67.42 gm-cm cm^{-2} in April, 1978. This represents an average decline of 1.54 gm-cm cm^{-2} per month during this period. Despite the rapid drop in meromictic stability between May and July, 1980, calculations for the next series of sampling dates (1980 to 1983) show a slower rate of decline, especially after July, 1980. Stability fell from 67.56 to 42.92 gm-cm cm^{-2}, an average

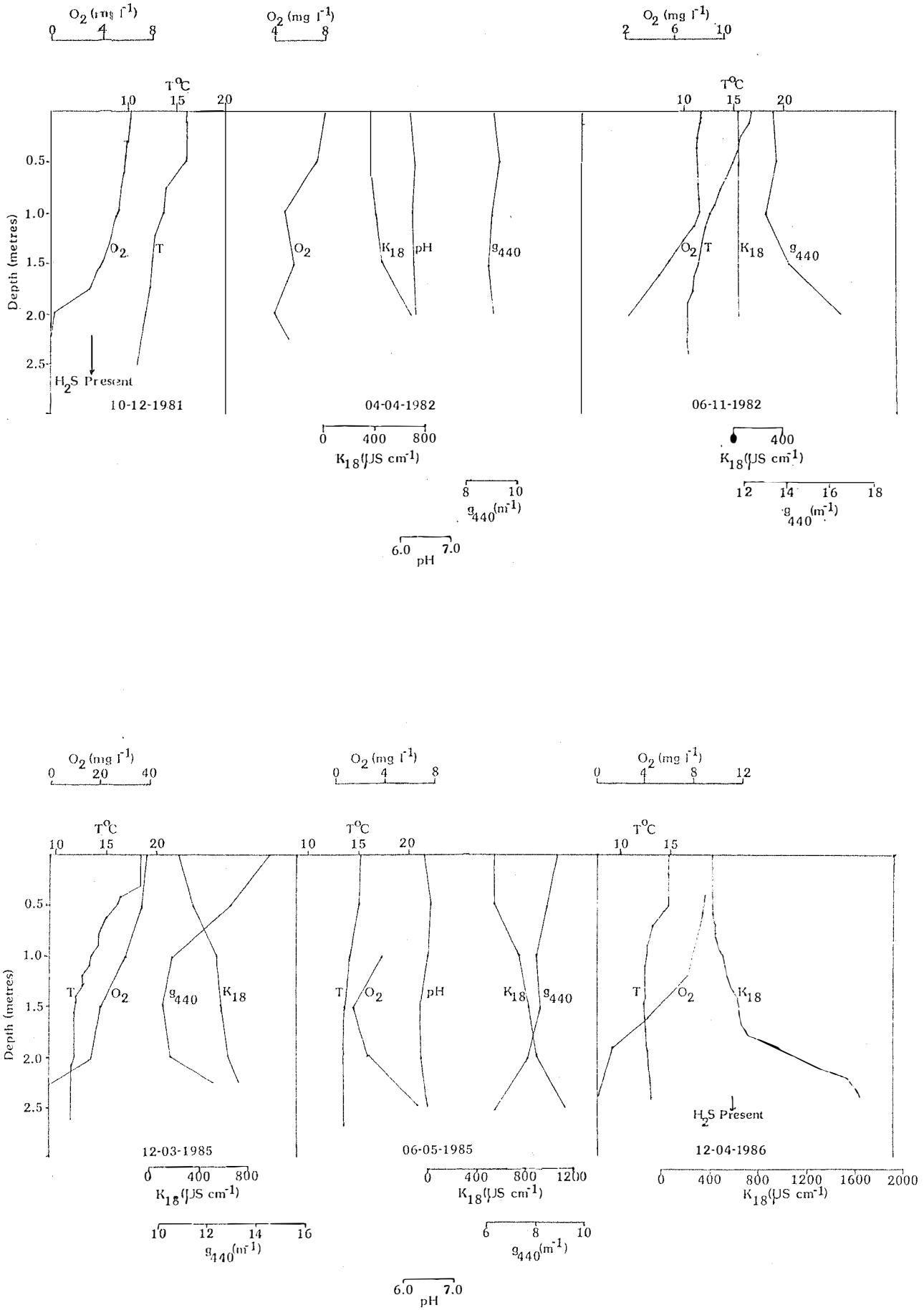


Figure 5.43 Depth profiles of various physico-chemical parameters from Lake Morrison.

decline of 0.70 gm-cm cm^{-2} per month. In the third set of data available (January, 1985 to April, 1987), meromictic stabilities dropped from 46.39 to 37.22 gm-cm cm^{-2}, an average of 0.33 gm-cm cm^{-2} per month. The lowest meromictic stability calculated, 28.32 gm-cm cm^{-2} in May, 1985, probably results from the higher salinity mixolimnion present on that occasion, which decreased the apparent density difference between this and the monimolimnion, rather than from other changes within the lake. Since 1986, meromictic stabilities here remained almost constant.

Meromictic stabilities in Sulphide Pool (Figure 5.45) were much lower than those of Lake Fidler. Stabilities declined gradually from a maximum of 9.44 gm-cm cm^{-2} in January 1977, to 6.02 gm-cm cm^{-2} in April 1978, a mean drop of 0.21 gm-cm cm^{-2} per month. The decline continued, from 4.13 gm-cm cm^{-2} in May 1980 to 0.07 gm-cm cm^{-2} in August 1983, an average decrease of 0.10 gm-cm cm^{-2} per month. Inflows of saline water on occasions since 1985 (5.3.11,above) have caused slight fluctuations in the chemical stabilities of the lake. However, apart from that of May, 1985, when the evidence suggests the greatest inflow, stabilities have always been below 1.00 gm-cm cm^{-2}.

The meromictic stability of Lake Morrison fluctuated markedly during 1977 and early 1978, prior to the onset of holomixis (Figure 5.46). These ranged from a maximum of 6.60 gm-cm cm^{-2} in June 1977, to a minimum of 2.10 gm-cm cm^{-2} in December 1977. Data from the following years reflect the periods when the lake is isohaline, with stabilities near zero, such as in July, 1980; May, 1981, and November 1982; and the intervening periods of incipient meromixis, when conductivity gradients were present. However, stabilities since 1978 have never exceeded 2.00 gm-cm cm^{-2}.

5.3.14 Thermal Stabilities, Birgean Wind Work, Heat Content, and Volume Weighed Average Temperatures

(a) Thermal stabilities

Thermal stabilities for Lake Fidler, Sulphide Pool, and Lake Morrison are shown in Figures 5.44, 5,45, and 5.46, respectively. These vary depending on the season and the extent of thermal stratification present. The highest thermal stabilities occurred in summer, when they could exceed the corresponding meromictic stabilities of these months, and thus assist the maintenance of meromixis. This was particularly so in Lake Morrison and Sulphide Pool. In contrast, thermal stabilities were lowest in winter. In

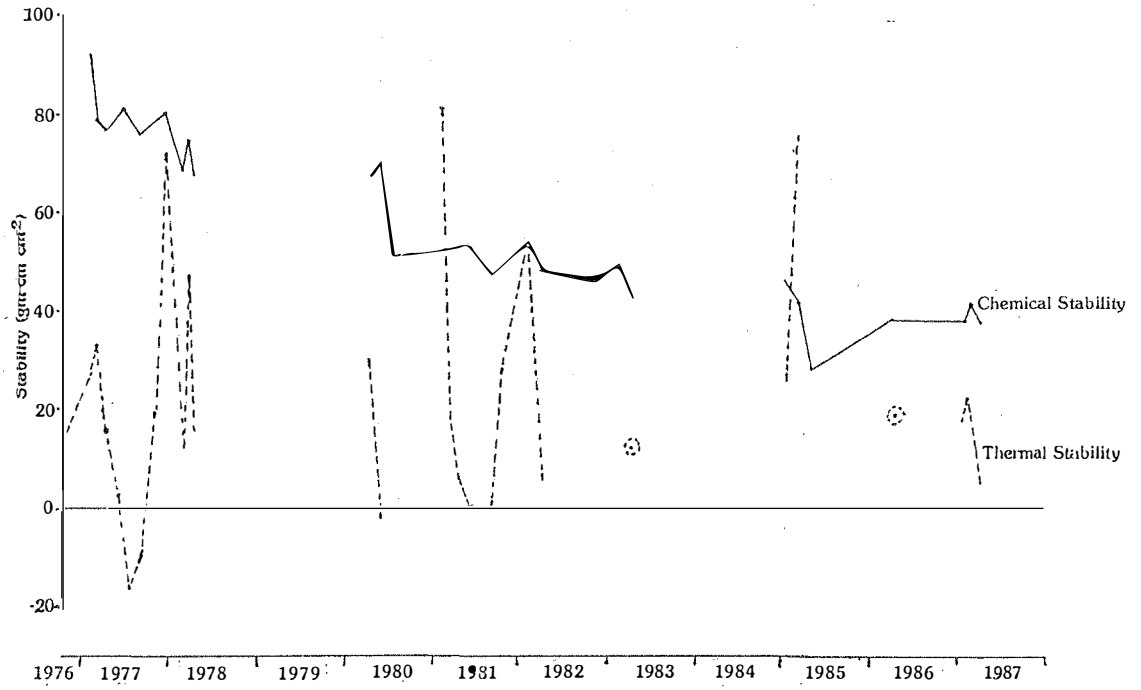


Figure 5.44 Chemical stabilities (solid line), and thermal stabilities (dashed line) calculated for Lake Fidler. Results prior to 1983 calculated from data of King (1980), Bowling (1981), Croome (1984), and Baker et al (1986a; and unpublished) .

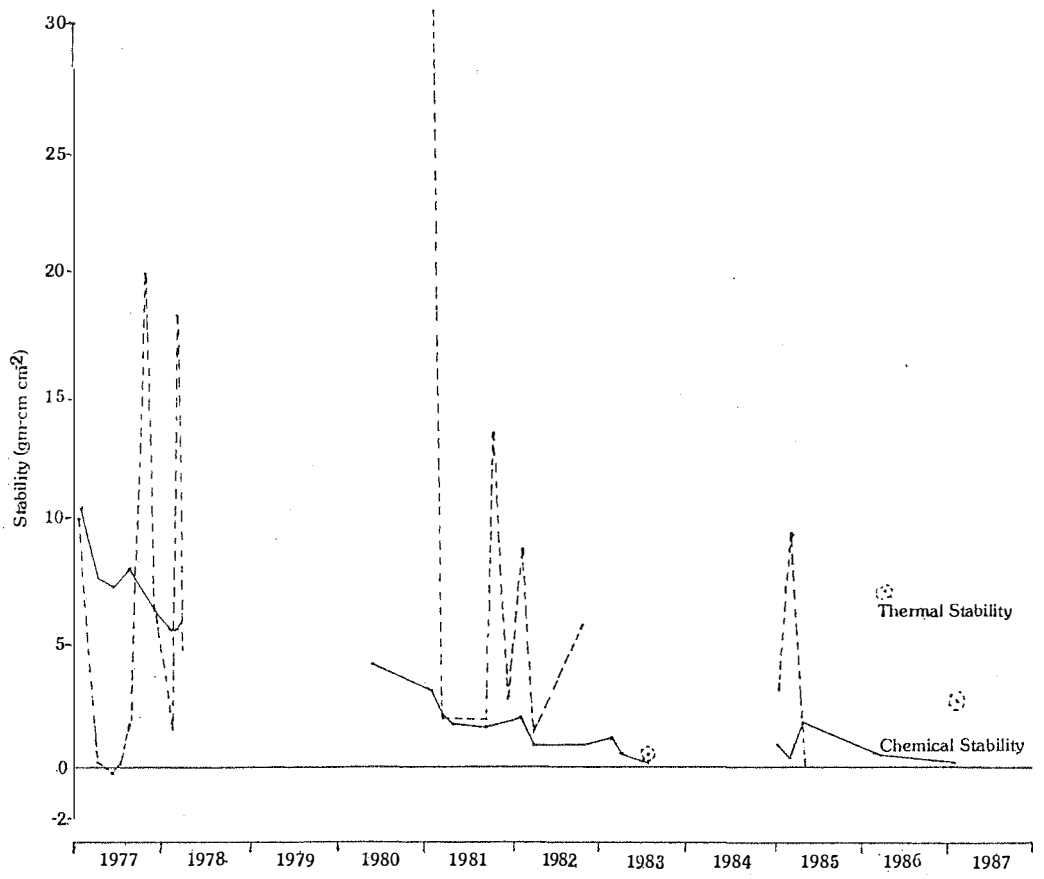


Figure 5.45 Chemical stabilities (solid line) and thermal stabilities (dashed line) calculated for Sulphide Pool. (Results prior to 1982 calculated from data of King (1980) and Bowling (1981))

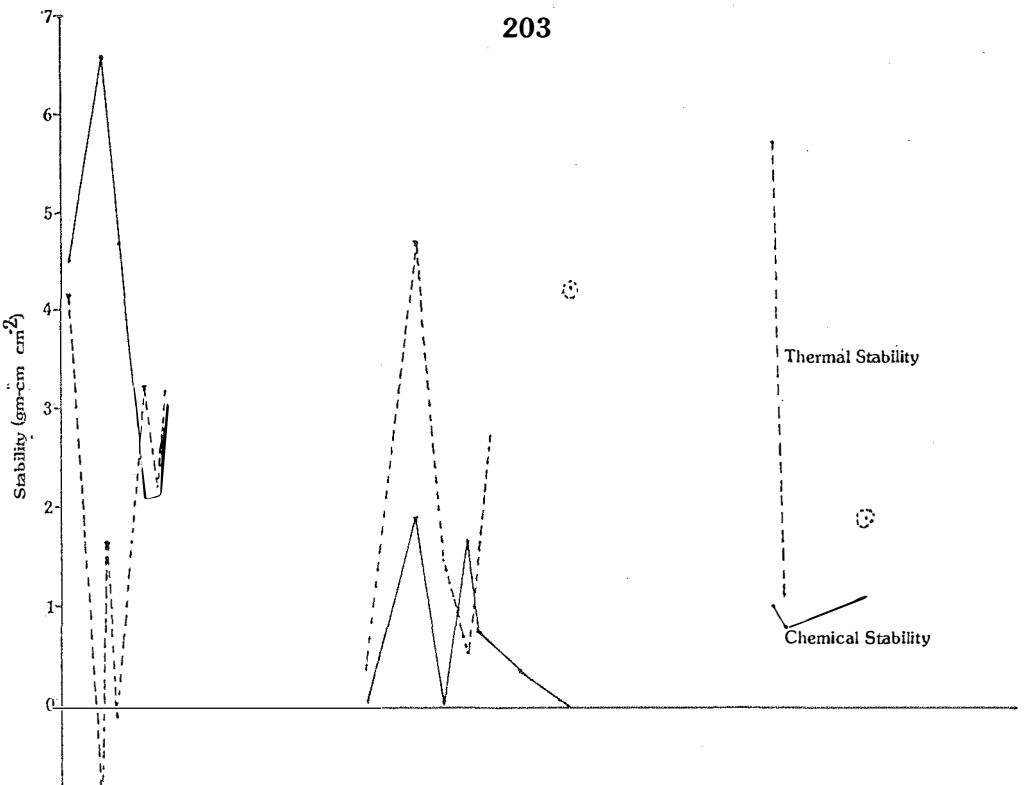


Figure 5.46 Chemical stabilities (solid line) and thermal stabilities (dashed line) calculated for Lake Morrison. (Results prior to 1982 calculated from data of King (1980) and Bowling (1981))

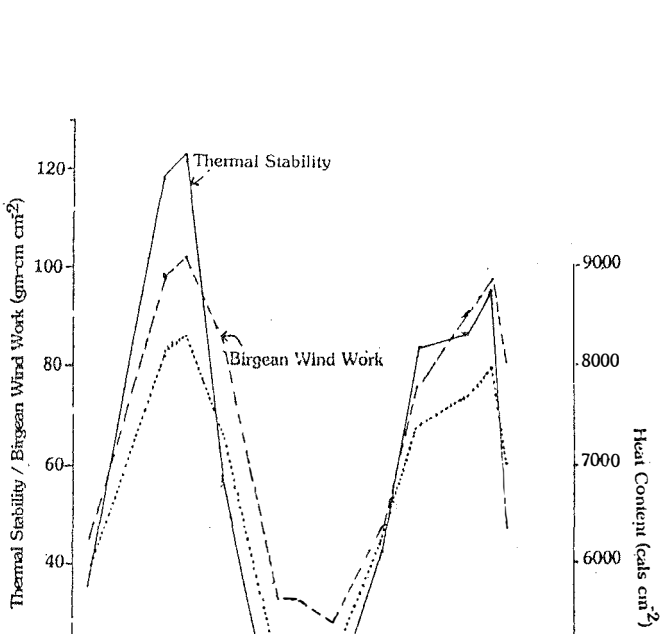


Figure 5.47 Thermal stabilities (solid line), Birgean Wind Work (dashed line), and Heat Content (dotted line) calculated for Perched Lake from October, 1976 to April, 1978, from the raw data of King (1980).

1977, when marked dichothermal temperature profiles were measured, negative thermal stabilities were calculated for all three meromictic lakes. At times, however, the normal seasonal pattern was distorted with marked fluctuations in thermal stability. These would probably be due to periods of hotter or colder weather causing warmer or cooler lake surface waters, and stronger or weaker thermal gradients than usual. The ranges calculated are given for each lake in Table 5.6.

Thermal stabilities were also calculated for nearby warm monomictic and mesohumic Perched Lake, using King's (1980) data for 1977 and 1978. The range in thermal stabilities is shown in Figure 5.47. The seasonal pattern of change was more regular than shown by the other three lakes, for the same period. Climatic differences probably account for the much lower stabilities of early 1978 than for the corresponding period of 1977.

(b) Birgean Wind Work

Birgean Wind Work values calculated for Lake Fidler, Sulphide Pool, and Lake Morrison are shown in Figures 5.48, 5.49, and 5.50, respectively, while those for Perched Lake from October, 1976 until April 1978, are shown in Figure 5.47. Ranges for the four lakes are given in Table 5.6. Birgean Wind Work generally showed smoother seasonal variations of less amplitude than thermal stability calculations, apart from the sudden increase in July 1977, in Lake Fidler, which possibly resulted from its severe dichothermal stratification at that time.

The values of Birgean Wind Work for Lake Fidler are lower from 1980 onwards (Figure 5.48). However, the data, especially from 1983 onwards, is too fragmentary to ascertain whether this is indicative of changes within the lake itself, or due to temporal differences. In comparison, recent values for both Sulphide Pool and Lake Morrison fall within the ranges calculated for these lakes for 1977 and 1978.

The most complete monthly data set for all four lakes was from January 1977 until April 1978. Direct work curves for selected months during this period are shown in Figure 5.51, and all results are shown plotted as isolines in Figure 5.52. Mesohumic Perched Lake exemplifies how work distributes energy in a warm monomictic lake during a year. In mid-summer, most energy is distributed within the surface two to four metres, but this distribution deepens in autumn, and during winter the little remaining energy is fairly evenly distributed

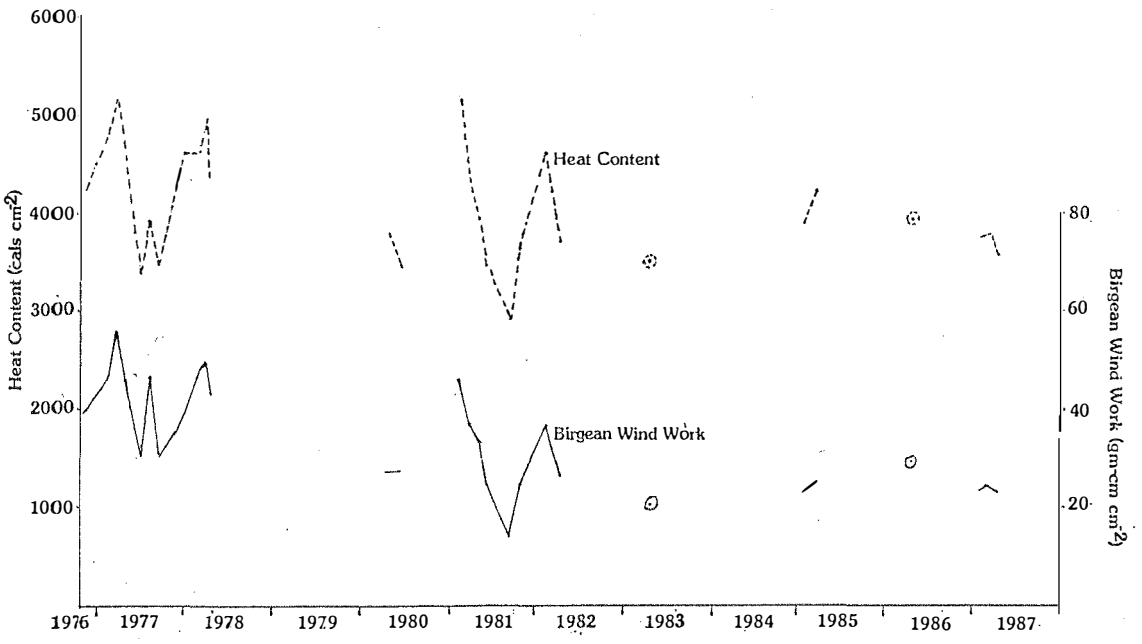


Figure 5.48 Heat content (dashed line) and Birgean Wind Work (solid line) values calculated for Lake Fidler. (Results prior to 1983 calculated using data of King (1980); Bowling (1981); Croome (1984), and Baker etal (1986a, and unpublished)).

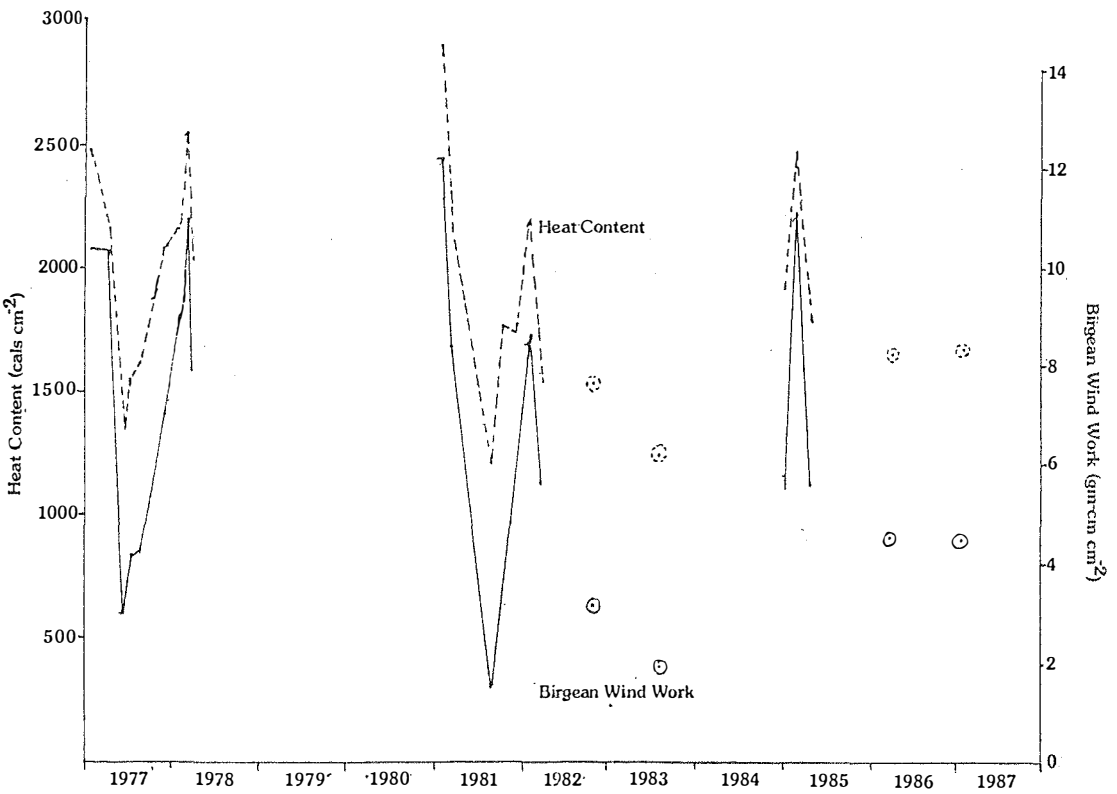


Figure 5.49 Heat content (dashed line) and Birgean Wind Work (solid line) values calculated for Sulphide Pool. (Results prior to 1982 calculated from data of King (1980) and Bowling (1981)).

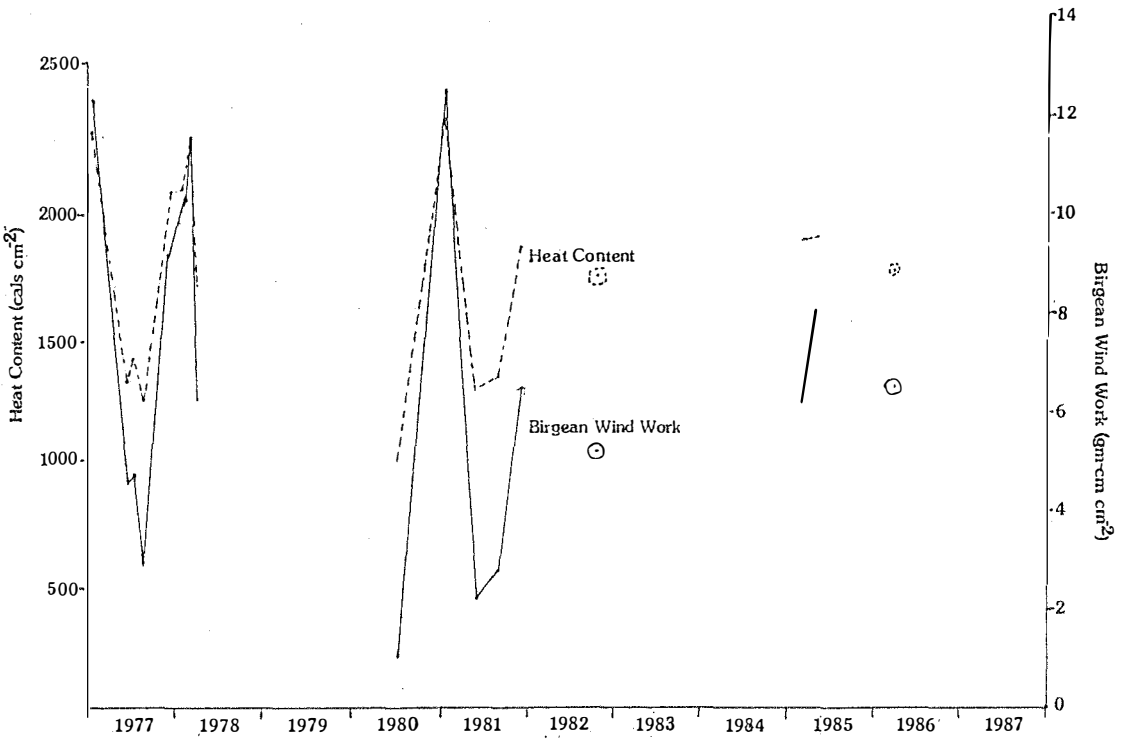


Figure 5.50 Heat content (dashed line) and Birgean Wind Work (solid line) values calculated for Lake Morrison. (Results prior to 1982 calculated from data of King (1980) and Bowling (1981)).

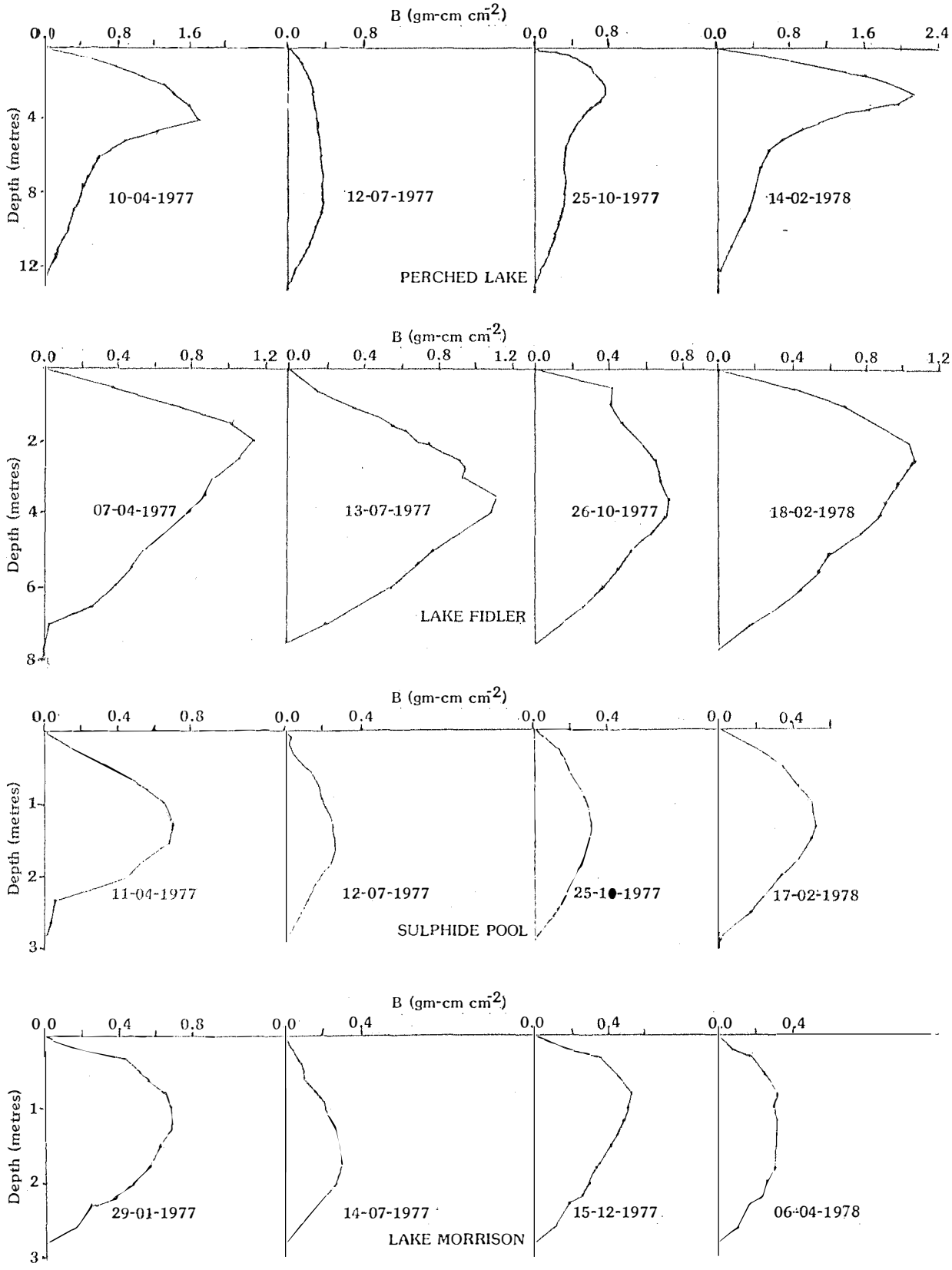


Figure 5.51 Direct work (B) curves for the four lakes of the lower Gordon River area, for summer, autumn, winter and spring periods, in 1977 and 1978 (calculated from King's (1980) raw data).

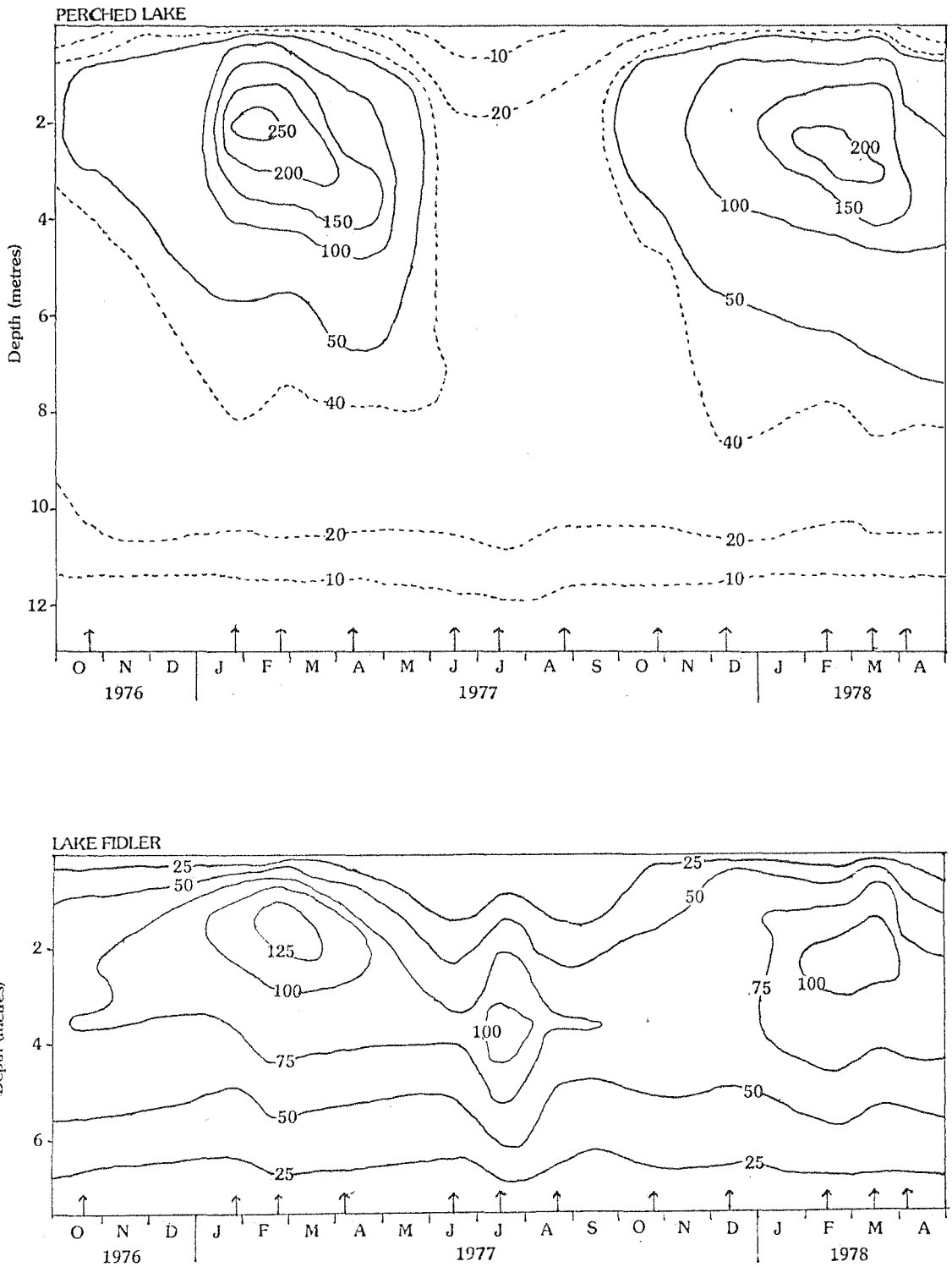


Figure 5.52 Isopleths of direct work (B) (gm-cm cm^{-2}) done by wind to distribute energy within a lake - Perched Lake and Lake Fidler, October 1976 to April 1978. Arrows along bottom indicate sampling occasions (calculated from raw data of King (1980)). (This figure continued next page.) B values $\times 10^{-2}$

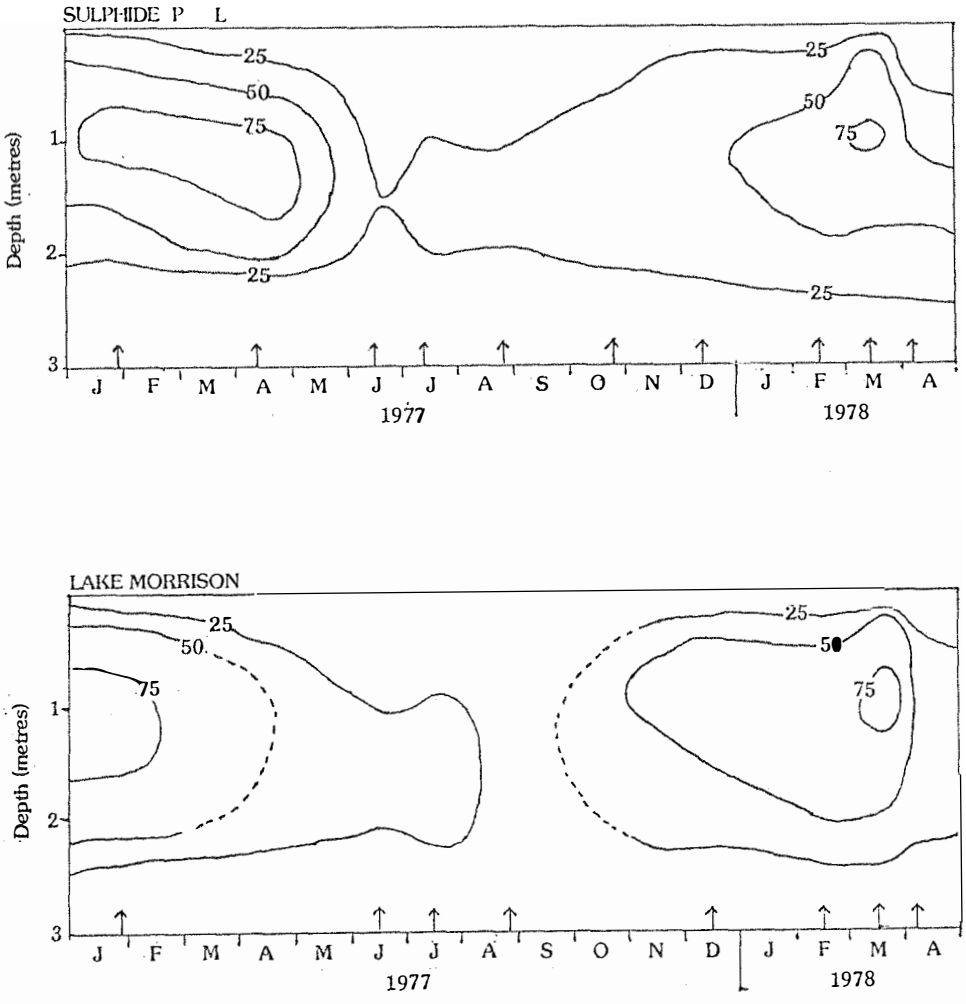


Figure 5.52 (continued) Isopleths of direct work (B) (gm-cm cm^{-2}) done by wind to distribute energy within a lake - Sulphide Pool and Lake Morrison. Arrow along bottom indicate sampling occasions. (Calculated from raw data of King (1980) . B values $\times 10^{-2}$

throughout all depths. The following spring wind work is again effective in distributing incoming solar energy only a little below the surface.

The direct work curves and isoline diagrams for the three meromictic lakes provide a considerable contrast to those of Perched Lake. Lake Fidler provides the best example. In autumn, maximum energy distribution is two metres below the surface of the lake, but in winter this drops to the mid-depths, due to the considerable amount of energy still remaining, stored as heat in the monimolimnion. The spring pattern, before extensive surface heating commences, is similar, but by summer most energy is again distributed near to the surface, a little above the chemocline. Similar patterns are also shown by both Sulphide Pool and Lake Morrison, but to a lesser extent.

(c) Heat Content and Volume Weighed Average Temperature

The heat contents calculated for the four lakes were also plotted in Figures 5.47, 5.48, 5.49, and 5.50, and the ranges given in Table 5.6. The marked seasonal changes in heat content are most apparent, being greatest during the late summer months, and least in winter. This allowed the estimation of annual heat budgets, θ_{ba}, for years when sufficient data were available. These were 1623.6, 1726.2, and 1720.0 cals cm^{-2} for Lake Fidler in 1977, 1980 and 1981, respectively; 1197.3 and 1011.4 cals cm^{-2} for Sulphide Pool for 1977 and 1981; and 1021.7 and 1378.5 cals cm^{-2} for Lake Morrison for 1977 and 1980. The heat budget for Perched Lake for 1977 was 3056.7 cals cm^{-2}.

Volume weighed average temperatures for the four lakes are listed in Table 5.7. Perched Lake was always colder than the meromictic lakes at the equivalent times of year, while summer average temperatures were generally lower in Lake Fidler than in either Lake Morrison or Sulphide Pool, but warmer during the winter. The average temperatures of Lake Morrison generally exceeded those of Sulphide Pool. The ranges recorded for the four lakes are given in Table 5.6.

5.4 DISCUSSION

5.4.1 The Role of Humics in the Limnology of these Lakes

Dissolved humic substances play a dominant role in the limnology of all these lakes, and contribute significantly to their meromictic or near-meromictic conditions.

Table 5.6: Range in Volume Weighed Average Temperature, Heat Content, Thermal Stabilities, and Birgean Wind Work for the four lakes of the lower Gordon River area. Dates of occurrence are included.

Lake	Volume Weighed Average Temperature °C		Heat Content Cals cm^{-2}		Thermal Stability gm-cm cm^{-2}		Birgean Wind Work gm-cm cm^{-2}	
	Max.	Min.	Max.	Min.	Max.	Min.	Max.	Min.
Fidler	16.1 (27-02-77)	9.0 (02-09-77)	5189.6 (27-02-77)	2909.6 (02-09-81)	81.11 (31-01-81)	-16.58 (13-07-77)	55.41 (27-02-77)	14.37 (02-09-81)
Sulphide Pool	18.4 (03-02-81)	7.6 (03-09-81)	2910.5 (03-02-81)	1199.1 (03-09-81)	30.56 (03-02-81)	-0.21 (16-06-77)	12.21 (03-02-81)	1.42 (03-09-81)
Morrison	17.5 (29-01-77)	7.6 (10-07-80)	2386.5 (30-01-81)	1008.0 (10-07-80)	5.71 (12-03-85)	-0.92 (17-06-77)	12.54 (29-01-77)	1.01 (10-07-80)
Perched	14.2 (23-02-77)	8.2 (12-07-77)	8273.6 (23-02-77)	4900.5 (12-07-77)	122.98 (23-02-77)	3.47 (15-06-77)	101.39 (23-02-77)	27.74 (24-08-77)

Table 5.7: Volume Weighed Average Temperatures (°C) for the lakes of the lower Gordon River

Date	Perched Lake	Lake Fidler	Sulphide Pool	Lake Morrison
October, 1976	10.3	13.2		
January, 1977	14.0	14.9	15.6	17.5
February, 1977	14.2	16.1		
April, 1977	12.4	14.1	13.6	
June, 1977	8.3	10.5	8.5	10.0
July, 1977	8.2	12.2	9.8	10.8
August, 1977	8.4	10.7	10.2	9.4
October, 1977	10.5	12.8	12.1	
December, 1977	12.7	14.4	13.1	15.7
February, 1978	13.3	14.3	13.8	15.9
March, 1978	13.7	15.5	16.1	17.0
April, 1978	12.1	13.6	12.8	12.9
April, 1980		11.7		
May, 1980		10.7		
July, 1980				7.6
January, 1981		16.0	18.4	17.9
March, 1981		12.3	13.5	
April, 1981		12.2		
May, 1981		10.7		9.7
September, 1981		9.0	7.6	10.0
October, 1981		11.5	11.1	
December, 1981			10.9	14.0
February, 1982		14.4	13.9	
April, 1982		11.5	9.6	
November, 1982			9.7	13.2
April, 1983		10.9		
August, 1983			7.8	
January, 1985		12.0	12.1	
March, 1985		13.1	15.5	14.2
May, 1985			11.2	14.4
April, 1986		12.3	10.4	13.4
early February, 1987		11.7	10.5	
late February, 1987		11.8		
April, 1987		11.0		

Dystrophic waters rapidly absorb solar radiation (Chapter Two), most of which is extinguished within the surface few metres of these lakes. This produces sharp, shallow, thermoclines which develop early in spring and persist well into autumn. This is particularly so in Lake Chisholm, where winter circulation lasts only three months; a much shorter period than in other Tasmanian lakes, such as the deep, exposed Risdon Brook Dam (Tyler, 1974), or even sheltered, mesohumic Perched Lake, which stratifies for between seven and eight months of the year (King and Tyler, 1981b). Jones and Arvola (1984) have suggested a close inverse relationship exists between water colour and the depth of mixing in small, sheltered, polyhumic forest lakes. The high humus concentrations may possibly allow weak surface heating even in winter. Temporary thermoclines with gradients of up to 5°C can develop even on clear, calm winter days in the Gordon River lakes (Baker et al, 1985a), creating further resistance to circulation.

The strong thermal stratification resulting from dystrophy is important in protecting the chemoclines and monimolimnions of the meromictic lakes and reservoirs from the erosive effects of circulation. This was shown particularly by Lakes Mackintosh, Rosebery, and Pieman, where chemical stratification disappeared once the lakes became isothermal in their first winter. The major erosion of the chemoclines and temporary salinity gradients of Lake Morrison and Sulphide Pool also takes place in the absence of thermal stratification, in winter. Prevention of circulation, either by thermal stratification or by ice cover, was an important cause of meromixis in Czech reservoirs (Fiala, 1979). Additionally, thermal gradients may impart considerable stability upon a lake, which in summer may exceed that produced by any salinity gradient present, and thus significantly assist the development and maintenance of meromixis.

A further effect of the long periods of thermal stratification in dystrophic lakes is hypolimnetic anoxia. This results from the bacterial decomposition of allochthonous humic substances rather than from autochthonous production (Salonen, 1981; Salonen et al, 1983). Hypolimnetic anoxia may develop very rapidly after the onset of thermal stratification, as in Finnish polyhumic lakes (Salonen, 1984; Salonen, Arvola, and Rask, 1984; Jones and Arvola, 1984), although several weeks elapse before it occurs in Lake Chisholm and the Pieman River reservoirs. A large volume, anoxic hypolimnion, present for considerable periods, may cause the reduction and mobilization of

solutes, and their accumulation in the bottom waters, leading to meromixis, as happened in the Pieman River reservoirs, Lake Barrington (Tyler and Buckney, 1974; Tyler, 1980), and Klíčava Reservoir, Czechoslovakia (Fiala, 1979).

Dissolved humic substances also lower the pH of the lakes and reservoirs studied, as they do in humic coastal lakes (Chapters Three and Four). Lake Chisholm waters are acidic, as were the surface waters of the Pieman River reservoirs, and Lake Barrington, and those of the Gordon River lakes (King and Tyler, 1981a,b; 1982a, 1983). This is in keeping with other humic Tasmanian and Finnish waters (Buckney and Tyler, 1973a,b; Arvola, 1985; Ilmavirta, 1980, 1983; Ilmavirta et al, 1984). The acidic nature of the Lake Chisholm waters, and its seawater ionic character, is unusual for a lake formed in a karstic depression. One explanation may be that the inflowing and resident waters are isolated from rock contact by the peat mantle of the forest floor and organic sediments of the lake bed. However, chelation and co-precipitation of calcium with humic substances (Sholkovitz and Copeland, 1981; Otsuki and Wetzel, 1973), and the conversion of bicarbonate ions to free carbon dioxide at low pH (Wetzel, 1975) such as those created by dissolved humic substances, may also account for the low concentrations of these ions. If so, this may prevent a build-up of these ions in the bottom waters of this lake, and biogenic meromixis by these ions.

Humic substances may also play a role in the redox equilibria of the monimolimnetic pools of chemically stratified Tasmanian reservoirs. Dickman and Hartman (1979) proposed that biogenically meromictic lakes could be subdivided into two mutually exclusive categories; those with an accumulation of bicarbonate or carbonate ions, and those with an accumulation of iron and manganese, in solution in their profundial waters. However, both metals, plus bicarbonate and sulphides, were present in solution in the monimolimnetic pools of the Tasmanian reservoirs of this study, without obvious precipitation of metal sulphides or carbonates. This may in part be due to the slightly acidic nature of these waters, caused by the presence of humic substances. However, a similar situation occurs in the anoxic hypolimnetic waters of dimictic Lake Lövösundet, Sweden, which Lidén (1983) characterized as a multicomponent system dominated by H^+, e^-, Fe^{2+}, Mn^{2+}, H_2S, H_2CO_3, and humic substances. He suggested a pH dependant complexation of the metals with the humic substances allows

the coexistance of all these components at a redox equilibrium. A like redox chemistry is hypothesised for the Tasmanian reservoirs. Additionally, humic substances also increase the solubility of certain metal ions, including iron and manganese, even in the presence of sulphides and carbonates (Rashid and Leonard, 1973).

5.4.2 The Influences of Basin Morphometry, Alignment, and Shelter from Wind Action

These three factors exert a considerable influence on the limnology of some of the lakes and reservoirs of this study, and their importance as aids to meromixis have been stressed in studies elsewhere (Northcote and Halsey, 1969; Weimer and Lee, 1973; Walker, 1974; Walker and Likens, 1975; Culver, 1975, 1977; Hongve, 1980; Howell and Kerekes, 1982).

Lake Chisholm is sheltered from most wind by a natural ampitheatre of forested hills, and is small and of considerable depth for its surface area. Both features impede circulation and favour stability (Walker and Likens, 1975), so that turbulant mixing is weak. Solar radiation absorbed by the surface waters is not distributed deeper within the lake by wind action, leading to a shallow epilimnion. Additionally, thermal stratification can develop early in spring and persist late into autumn, without destruction by wind created turbulence. The roles of basin morphometry, and of shelter provided by surrounding hills and forests, have also been recognized as being limnologically significant in many Finnish polyhumic forest lakes similar to Lake Chisholm (Salonen, Arvola and Rask, 1984; Arvola, 1983, 1984b).

All four Pieman River reservoirs showed initial chemical stratification, but this persisted beyond the first summer only in Lake Murchison. This results from differences in their basin morphometry, alignment and shelter (Figure 5.3a). Lake Murchison is long, narrow, and sinuous, well protected by surrounding hills and mountains, especially near the dam site, all of which reduce wind mixing to a minimum. In comparison, Lake Mackintosh is much broader, especially in the arm occupying the valley of the Sophia River (Peterson and Missen, 1979), and therefore much more prone to wind action. Lakes Rosebery and Pieman are also broader and more exposed than Lake Murchison, and their west-east alignment would make them more susceptable to mixing by the prevailing westerly winds acting

over their entire lengths. Thus depth, shelter, and the dendritic nature of Lake Murchison, which limits wind fetch, all allowed meromixis to remain in this reservoir, as they did in Lake Barrington (Tyler and Buckney, 1974; Tyler, 1980).

The low meromictic stabilities calculated for the meromictic Gordon River lakes point to the fragile nature of this phenomenon within them, which, compared with stability results from meromictic lakes elsewhere (Table 5.8), are more akin to biogenic rather than ectogenic meromixis, especially those of Lake Morrison and Sulphide Pool. Campbell and Torgersen (1980) have described meromictic stabilities of 5.5 to 9.5 gm-cm cm^{-2} as "rather precarious". Factors other than the salinity gradients, such as basin morphometry and shelter, are therefore indicated as being operative, and although insufficient to fully maintain meromixis, may aid to slow its rate of decline.

The different rates at which the three meromictic Gordon River lakes have proceeded towards holomixis result principally from differences in basin morphometry, described in detail by King and Tyler (1981a), and shelter. Lake Fidler, though sheltered, is large enough (1.28 ha) to permit wind generated ripples, but is deep enough (7.6 metres) to prevent overturn. Its steepsided basin morphometry, plus its protected nature, has meant this lake proceeded more slowly than the others towards holomixis. As the chemocline sinks, it is possible that the resistance offered to mixing by its basin shape and the salinity density gradient may approach an equilibrium with the circulatory effects of the weak wind action on the surface waters. In comparison, although highly sheltered and small (0.11 ha), Sulphide Pool moved more rapidly towards holomixis, so that by August, 1983, only a shallow monimolimnion remained. Subsequent saline gradients created by more recent inflows also appear to be eroded quickly. The shallowness (2.7 metres) of Sulphide Pool allows even the small amount of wind induced circulation to be effective, in the absence of a strong salinity gradient, over almost its entire depth. Lake Morrison is of similar depth to Sulphide Pool, but of similar area to Lake Fidler, and is also the most exposed. It quickly became holomictic once river regulation began (King and Tyler, 1982b, 1983), and periods of incipient meromixis since have also been rapidly destroyed. Thus, the favourable basin morphometry and shelter of Lake Fidler allows only sluggish circulation there,

Table 5.8: Meromictic Stabilities of the Gordon River meromictic lakes, compared with those for selected meromictic lakes elsewhere

Lake	Stability gm-cm cm^{-2}	Reference
Lake Fidler	92-43	This thesis
Lake Morrison	6.6-0.01	This thesis
Sulphide Pool	9.4-0.07	This thesis
Ectogenic/Crenogenic Meromictic Lakes		
Six Lakes in Central Washington, U.S.A.	410-4670	Walker, 1974
Big Soda Lake, Nevada, U.S.A	91,500-49,400	Kimmel et al, 1978
West Basin Lake, Victoria, Australia	878	Timms, 1972
Fayetteville Green Lake, New York, U.S.A.	1700	Brunskill and Ludlam, 1969
Various Antarctic meromictic lakes	553-2609	Burton, 1981
Laytons Lake, Nova Scotia, Canada	88-124	Howell and Kerekes, 1982
Biogenic Meromictic Lakes		
Lake Mary, Wisconsin, U.S.A	1.1	Weimer and Lee, 1973
Hall Lake, Washington, U.S.A.	1.65	Culver, 1977
Lake 120, Ontario, Canada	5.5-9.5	Campbell and Torgerson, 1980
Eight lakes in the Romericke district, Norway	0.01-18.4	Hongve, 1980

while the area, shallowness, and exposure of Lake Morrison freely enables mixing to all depths. Sulphide Pool lies somewhere between these two extremes.

5.4.3 Heating and Mixing Dynamics

The annual heat budgets of Lake Chisholm and the Gordon River lakes are small, and although those of the Pieman River reservoirs were higher, these too were small for lakes of their size (Gorham, 1964). These give some quantification of the influences of shelter on these lakes, suggesting that ~~little~~ work has been done by the wind to distribute heat within them. This small amount of heat, coupled with the high humus content trapping most heat close to the surface, means that the capacity of the lakes to store heat is greatly reduced, as only the epilimnetic or mixolimnetic waters participate in heat exchange with the environment, and thus in the determination of annual heat budgets. The heat stored in the monimolimnions of the meromictic lakes would also have only limited participation. Additionally, because most heat is absorbed close to the surface, it can be quickly lost back to the atmosphere at night, rather than being mixed to warm the deeper waters of these lakes, and thus retained. Large diurnal temperature fluctuations occur in the surface waters of the Gordon River lakes (King and Tyler, 1982a; Baker et al, 1985a; Bowling, 1981). Similar low annual heat budgets have also been reported for eight small, protected Norwegian meromictic lakes (Hongve, 1980), and for a sheltered biogenically meromictic lake in Washington (Culver, 1977).

The values of annual heat budgets are dependant on both the size and depth of a lake, increasing as lakes become larger or deeper, but at a lessening rate (Gorham, 1964). Thus, the annual heat budgets of the Pieman River reservoirs were highest, while those of Lake Chisholm and the Gordon River lakes were considerably less. The calculations are also dependent on the timing of the thermal measurements used, as the heat contents of lakes may vary significantly over a period of just two or three days, making the specific time of maximum heat content unpredictable (Stewart, 1973). This would also be true for the minimum heat content in warm monomictic lakes circulating above 4°C in winter. With a sampling pattern of one measurement per month, or at even longer intervals, the periods of maximum and minimum heat contents of these lakes could very easily have been missed, leading to under-estimations of their annual heat budgets. Even the time of

day of measurement may have some effect, if large diurnal temperature changes occur.

Because heat content is a measure of the amount of heat per unit surface area, between lake comparisons cannot easily be made, due to differences in depth. However, volume weighed average temperatures allow this, being the equivalent of the heat content per unit volume, calories cm^{-3} (Stewart, 1973). Warm monomictic Perched Lake, Lake Chisholm, and the Pieman River reservoirs were always cooler than the Gordon River meromictic lakes, possibly because shallow thermal stratification leaves a considerable volume of cold hypolimnetic water which reduces their average summer temperatures, while they also lack warm, stagnant, monimolimnetic waters which could elevate average winter temperatures in the Gordon River lakes. The shallowness of both Sulphide Pool and Lake Morrison may have allowed greater summer heating, and winter cooling, of the entire water columns of these lakes, so they were generally warmer in summer and colder in winter compared to Lake Fidler. The larger surface area and exposure of Lake Morrison may also have allowed greater opportunities for wind mixing to distribute heat more equitably with depth, thus raising its average summer temperature. Gorham (1964) considered this the main function of increasing lake area causing higher annual heat budgets. Since 1978, Sulphide Pool and Lake Morrison have shown greater extremes than previously, but this may be due to temporal differences between years, rather than an effect of the demise of their meromixis in more recent years. Temporal differences are apparent in Lakes Chisholm and Murchison, too.

Birgean Wind Work may be used to show the exposure of a lake to wind, and is the minimum amount of work required to distribute a given heat load from an initial condition (usually assumed to be an unstratified lake at 4.0°C) to produce the observed density stratification (Idso, 1973). Low values indicate a predisposition towards meromixis (Hongve, 1980). The values for Lake Chisholm are very low, even in summer, as are those for the Gordon River lakes, while warm monomictic Perched Lake had the highest values. These also fall within the same order of magnitude as those reported by Hongve (1980) for highly stratified Norwegian lakes, and result from wind action having very little effect in distributing heat from the surface to the deeper parts of the lakes. Additionally, the calculation does not differentiate between heat actually mixed

downwards by wind action and that generated at each depth by direct solar heating, which may account for as much as 50% of the heat distribution in well protected lakes (Hongve, 1980). Also, calculations for meromictic Lakes Fidler, Morrison and Sulphide Pool may be influenced by the amounts of heat stored within their monimolimnions, and not included in the mixing processes. Because of this, Birgean Wind Work may be of limited value only in describing the mixing dynamics of such lakes. Wind Work values calculated for the Pieman River reservoirs and Lake Barrington were higher than for the other lakes studied, but those for Lake Murchison especially were low for lakes of this size and depth, indicating limited wind mixing there as well.

Thermal stabilities are estimates of the amount of work required to mix a thermally stratified lake to a new, uniform, isothermal condition, without gain or loss of heat (Idso, 1973). Those of Lake Chisholm and the Gordon River lakes correspond well with those reported for Norwegian lakes by Hongve (1980), but are much lower than those calculated for the Pieman River reservoirs and Lake Barrington. However, they are still sufficient to impart considerable stability to the water column during periods of thermal stratification, thus aiding the development and maintenance of meromictic or near-meromictic conditions in the lakes.

5.4.4 Meromictic Tendancies in the Lakes and Reservoirs

The warm monomictic stratification cycle of Lake Chisholm differs markedly from the dimictic conditions prevailing in Finnish polyhumic lakes, but the effect of their humus content on heat absorption in early spring is the same. With rapid heating of their surface waters, many Finnish lakes transform swiftly from ice cover (which prevents winter mixing), to spring stratification, before full ventilation of the hypolimnion has been achieved; a condition known as "spring meromixis" (Arvola and Rask, 1984; Salonen, Arvola, and Rask, 1984). Dystrophy also creates a tendancy towards incipient meromixis in Lake Chisholm, evidenced by the lingering autumnal and early spring stratification, and even in winter circulation throughout the entire lake is sluggish, and oxygen is distributed homogeneously for only four to six weeks during July and August. Despite changes in major solute concentrations across the thermocline being minor, the water column none-the-less has considerable physiochemical structure, particularly with respect to temperature, oxygen, redox chemistry, and light. Thus, Lake

Chisholm closely resembles a Type IV meromictic lake of Walker and Likens (1975).

There is little enrichment of the hypolimnetic waters of Lake Chisholm, despite the long period available for mineral accretion during thermal stratification. The scarcity of iron and manganese is notable. Given the long period of anoxia, it is hypothesised that, had these elements been present in the bedrock of the lake, sufficient concentrations would be reduced and brought into solution to create density differences sufficient to resist the weak circulation and render the lake permanently biogenically meromictic in the manner of some Norwegian lakes (Kjensmo, 1968; Hongve, 1980). Also, biological production is apparently too small for ions from biological sources to accumulate during thermal stratification and cause meromixis in the manner of Lake Mary (Weimer and Lee, 1973), Sunfish Lake (Duthie and Carter, 1970) and Hemlock Lake (Fast and Tyler, 1981). Instead, the near meromictic condition of Lake Chisholm is entirely due to its morphometry and shelter, so that it tends towards "morphogenic meromixis" as described by Northcote and Halsey (1969).

In keeping with Tyler's (1980) prediction, chemical stratification formed in all four Pieman River reservoirs, although this was only a fleeting episode early in the development of Lakes Mackintosh, Rosebery, and Pieman. However, it persisted throughout the study in Lake Murchison, although a gradual decline was evident from mid 1984 onwards. The meromixis was clearly of biogenic origin, with the monimolimnetic waters dominated by calcium, magnesium, and bicarbonate, along with iron and manganese. These are typical of biogenically meromictic lakes (Dickman and Hartman, 1979). Other causes of meromixis, such as triptogenesis, as occured in Hills Creek Reservoir, Oregon (Larson, 1979), or annual saline inflows like those maintaining ectogenic meromixis in Lake Powell (Johnson and Merritt, 1979) are improbable, as water entering the Pieman River reservoirs is both dilute and non-turbid.

Chemical stratification evolved quite rapidly after the formation of each reservoir. It was incipient in Lakes Rosebery and Pieman only three to four months after their creation, when increases in pH, K_18, and alkaline earth bicarbonates were evident in their bottom waters. The early onset of meromixis in Lake Murchison was less marked, with only slight increases in K_18, and temperature near the bottom.

Meromixis developed further during the first summer of impoundment, after hypolimnetic anoxia brought about by decaying vegetation from the newly flooded river valleys under conditions of thermal stratification allowed the reduction and accumulation of dissolved iron and manganese at the base of the dam. Lake Mackintosh too was meromictic seven months after its formation. These processes leading to the early biogenic meromixis of the Pieman River reservoirs closely follow those outlined by Tyler (1980).

All the reservoirs had high level offtakes (Figure 3b), leaving a considerable depth of dead water below them for the establishment of chemical gradients. The onset of triptogenic meromixis in Hills Creek Reservoir was due partly to the dam lacking low level outlets to purge highly tubid bottom waters (Larson, 1979), as occurred in Lake Rowallan, Tasmania, where drawoff via low level outlets destroyed incipient biogenic meromixis (Tyler and Buckney, 1974). Withdrawal through high-level outlets also creates convective effects which aid the establishment of meromixis in two ways. Firstly, the extraction via such outlets create extensive withdrawal currents at depths corresponding to the outlet, removing the surface waters but leaving the bottom waters relatively undisturbed (Fiala, 1965; Johnstone and Merritt, 1979; Welsh, 1984). Secondly, convective circulation below these depths is blocked, further protecting the monimolimnetic pool. This effect occurs in both Lake Powell, U.S.A. (Johnstone and Merritt, 1979), and in Dartmouth Dam, Victoria (Welsh, 1984).

Three possible methods for the initiation and maintenance of meromixis in the Gordon River lakes were proposed by King and Tyler (1981a); firstly, that the salts were relictual, trapped in the bottom waters after levee bank deposition; secondly, that the salts entered the lakes by percolation of salt wedge waters through the levee banks; and thirdly, saline waters could occasionally flow into the lakes via creeks connecting them with the river.

The investigations reported here provide strong support for the third alternative. The rapid demise and eventual elimination of meromixis from Lake Morrison and possibly Sulphide Pool since 1977 militates against their brackish monimolimnia being relictual. Fossil pigment analysis of Lake Morrison sediments (Bowling, 1981) indicate the lake had been meromictic for a long period of time, so clearly some form of maintenance of meromixis by salt replenishment

is required. Limited data on ground water from bores along the river bank (Kearsley, 1978) show rapid declines in sodium and chloride concentrations with distance from the river, so the second alternative is also unlikely.

The third theory, that turbulent mixing raises the surface salinity of the river whilst the salt wedge is present, and that this then enters the lakes via the connecting creeks, also warrants examination. Because they are infrequent events, no actual inflows into the lakes have been observed, but the alternation of holomixis with periods of incipient meromixis in Lake Morrison, and similar inflows creating new conductivity gradients in Sulphide Pool, leaves little doubt as to their occurrence. Inflows also occur in Lake Fidler, but are too dilute to affect the monimolimnion, although they may replace the mixolimnion instead. The marked conductivity gradients present in both Lake Morrison and Sulphide Pool, and the considerably elevated mixolimnetic conductivities in Lake Fidler, in May, 1985, indicate simultaneous saline inflows into all three just prior to this date. The creeks have sufficient capacity to allow the rapid exchange of large volumes of water between the lakes and the river (Bowling. 1981). Coastal meromictic lakes on Åland are also maintained by similar occasional incursions of marine salts from the Baltic Sea (Lindholm, 1975, 1982), as is meromixis in the Swartvlei estuary, South Africa (Allanson and Howard-Williams, 1984).

5.4.5 The Future of Meromixis in Lake Murchison, Lake Barrington, and the Gordon River Lakes

Although meromixis still persisted in Lake Murchison three years after the formation of this reservoir, it appeared to be in decline, especially during 1985, when the chemocline sank below 60 metres. Additional wind induced circulation that year may have caused the erosion of the upper layers of the monimolimnetic pool. Climatic data from Rosebery, the nearest meteorological station, indicate 1985 was windier than either 1983 or 1984, having eight days of strong wind (mean wind speed in excess of 21 knots), compared to only one in 1984, and zero in 1983 (Data from Commonwealth Bureau of Meteorology, Hobart).

Some shaving of the monimolimnion by cold density currents flowing over it, such as observed in July, 1984, may also have occurred, increasing the depth of the chemocline, and removing solutes. Cold density currents may be frequent within the reservoir

during winter. Water at or close to maximum thermal density (4.0°C) was present in the mid-depths of the reservoir in July, 1983, and July 1985, overlying the warmer monimolimnetic waters and pointing to their presence on these occasions. Lake Murchison differs from Lake Barrington in this regard, where few cold density currents would exist to disrupt the monimolimnetic pool, due to this reservoir receiving homogenized water from another impoundment immediately upstream. However, cold density currents are insufficient by themselves to totally disrupt the meromixis of Lake Murchison. This contrasts with Lake Gordon, Tasmania, and Dartmouth Dam, Victoria (Steane and Tyler, 1982; Welsh, 1984), where cold density currents underflow and destroy any incipient monimolimnetic waters. Ferris (1985) reported cold inflows causing similar stratification behaviour in Lake Burragorang, New South Wales.

It is unlikely the decline of meromixis in Lake Murchison resulted from the exhaustion of organic materials remaining from the pre-flooded river valley, the decay of which would aid replenishment of the monimolimnetic pool. Anoxia in the lower hypolimnion in March and November 1985 (Figure 5.21) suggests sufficient organics remaining or produced within the reservoir for this not to be so.

The declining meromixis in Lake Murchison may not necessarily spell its end there. Similar events occurred in the years following the formation of Lake Barrington, where by 1978, the dramatic changes associated with meromixis had all but disappeared (Steane, unpublished data). Tyler (1981) predicted the end of meromixis in that reservoir. However in October 1982, Lake Barrington was again intensely chemically stratified, with several parameters being higher than ever recorded. This condition was still being maintained in October 1985 and the probability exists of its long-term presence. Similarily, meromixis may also remain in Lake Murchison.

The re-emergence of meromixis in Lake Barrington between 1978 and 1982 cannot be easily explained, although drought conditions in the early 1980's resulted in lower average daily flow rates through the reservoir (Hydro-Electric Commission of Tasmania, personal communication) which may have aided it. Fast and Tyler (1981) have shown how rapidly biogenic meromixis may become re-established in lakes under suitable conditions, and Fiala (1979) gave evidence of meteorological events causing the re-establishment of meromixis in Czechoslovak reservoirs.

Favourable circumstances must have developed prior to 1982 to trigger the re-development of meromixis in Lake Barrington.

Lake Fidler is the only one of the three Gordon River meromictic lakes not to have reverted to holomixis since the alteration of the flow regime of the river in 1977, although its meromixis, as measured by meromictic stability, has declined 50% since then. Lake Morrison quickly became holomictic in 1978 (King and Tyler, 1983), while meromixis in Sulphide Pool had declined by August, 1983, to a point where it was extremely tenuous. Whether holomixis eventuated is uncertain, as no further measurements were made for sixteen months, by which time new saline inflows had occurred, but a period of holomixis during this time is possible. However, any remnants of the original meromixis still present in 1983 would have been swept from the lake, and replaced by the newly created salinity gradients resulting from the more recent saline inflows.

Although salt wedge intrusions still occur in the Gordon River, the higher river flow rates now cause considerable dilution and reduced mixing with the river surface waters. Consequently, inflows of brackish waters into the lakes are now of insufficient concentration to create density gradients of appropriate magnitude to resist even the weak, wind-induced circulation. The salinities of the bottom waters of Lake Morrison during periods of incipient meromixis are only half or less of those formerly recorded there by King and Tyler (1983), as are salinities of recent inflows into Sulphide Pool. Further, salt wedge penetration is much less frequent than previously (Kearsley, 1978, 1982), so periods between salt replenishment in the lakes are longer.

The future of meromixis in the Gordon River lakes under the present river flow regime is now apparent. Lake Morrison should continue to vacillate between periods of holomixis and temporary ectogenic meromixis, as detailed previously (Bowling, 1981). Sulphide Pool now displays a similar limnological pattern of saline inflows, but the decay of the concentration gradients so created is considerably slower than in Lake Morrison. Thus, new gradients may be established by new inflows before the previous gradient has completely eroded and the lake rendered holomictic, although occasional holomixis should eventuate in the event of long periods without inflows. Inflows should be less frequent in Sulphide Pool than in Lake Morrison, due to its location further upstream. In comparison, the rate of demise

of meromixis in Lake Fidler has slowed. This notion is supported by the diffuse nature of the halocline, and by the lingering oxycline within the lower mixolimnion, indicating very weak mixing at these depths. If so, meromixis should persist for a long time yet in Lake Fidler, although continued loss of monimolimnetic salts will continue to weaken it. However, the chemocline may sink too deep to receive adequate light to ensure the continuance of the micro-organisms stratified about it. Parkin and Brock (1980a,b) have shown the necessity of sufficient light for the existance of bacterial plates in Wisconsin lakes.

5.4.6 Significance of the Gordon River Lakes to the World Heritage Area of South-west Tasmania

The Gordon River lakes have a special significance in that they lie within the World Heritage Area of south-west Tasmania, and their limnological properties serve to enhance the conservation value of the area (Tyler, 1986). Firstly, the backswamp lakes are excellent examples of meromixis, displaying pronounced salinity gradients accompanied by other abrupt chemical changes, at shallow depths (King and Tyler, 1981a, 1982a, 1983; Baker et al, 1985a). Meromixis itself is an unusual limnological phenomonena (Walker and Likens, 1975). The lakes also display unique biological features, including a highly stratified array of algal and bacterial micro-organisms straddling their chemoclines, and other newly described or undescribed species of micro-algae occur within their mixolimnetic waters, as well as in near by Perched Lake (Croome, 1986; Tyler, 1986). Additionally, the three meromictic lakes, with their surrounding marginal rafts of herbfields, or "Schwingmore", dramatically illustrate ongoing processes of terrestrialization (Tallis, 1973). Thus, they meet the World Heritage criteria (Mulvaney, 1983) as outstanding examples of ongoing processes of biological evolution, and of the development of freshwater bodies. Unfortunately, the means which have brought about the demise of their meromixis also serves to illustrate another World Heritage criterion; that of Man's interaction with his natural environment.

By chance, the lakes also have considerable cultural value. The Gordon River area has rich archaeological cave sites, indicating aboriginal habitation of south-west Tasmania during the last ice age, 15 to 20 Kyr ago, when they were the most southerly humans of those times (Kiernan et al, 1983). The lakes are the only ones in this

area, and their sediments offer a corroborative chronology of climatic and vegetative conditions in the area for at least part of the period of Aboriginal occupation. The special limnological properties of the meromictic lakes make them particularly suited sites for this. Meromixis, with its permanent stratification, accompanied by anoxia in the monimolimnion, means that the stratigraphy of the bottom sediments is undisturbed by water movements or by burrowing animals, so that they offer a microfossil chronology of high resolution. Although the meromictic lakes probably date only from the stabilization of sealevel at its present position, about 6 to 8 Kyr B.P., Perched Lake is probably much older, and would extend the chronology, albiet in a less precise manner, to even earlier periods. Currently the past climatic and vegetational history of the area must be extrapolated from the pollen records of highland areas of Central and Southern Tasmania (Macphail, 1975, 1979), which may differ greatly.

5.4.7 The Ecology of Phytoplankton in Lake Chisholm

While the Gordon River lakes are of considerable limnological significance, so too is Lake Chisholm, as it is the only polyhumic forest lake of its type in Tasmania, and it is also a rich phytoflagellate habitat, especially for chrysophytes.

The characteristic physiochemical features of polyhumic lakes create a difficult environment for phytoplankton, given the poor underwater light climates for photosynthesis, the limited mixing due to weak wind action and strong thermal stratification, the low pH, and the often low nutrient levels in the surface waters (Arvola, 1985). These conditions have been identified as favouring flagellates (Ilmavirta, 1980, 1982, 1983, 1984; Ilmavirta *et al.*, 1984), and some may contribute to their domination of Lake Chisholm. Flagellates also predominate in the small, brown-water forest lakes of Finland (Ilmavirta, 1983, 1984; Arvola, 1985, 1986), and in Lake Fidler (Croome and Tyler, 1986). The limited epilimnetic circulation may also explain the paucity of desmids and diatoms in Lake Chisholm.

The limited nutrient data from Lake Chisholm suggest that phosphorus is unlikely to be limiting for phytoplankton growth, and the lake could be regarded as mesotrophic or eutrophic (O.E.C.D., 1982). Nitrate concentrations in the lake are more indicative of low trophic status (Wetzel, 1975). Nutrient levels were of the same order of magnitude as those reported from Finland (Ilmavirta, 1980, 1983; Arvola, 1983, 1984a,

b; Rask *et al.*, 1986; Salonen *et al.*, 1983; Ilmavirta *et al.*, 1984), although orthophosphate was more readily available in the epilimnion, and changes in concentrations of nutrients across the thermocline were not as marked.

Marked seasonal succession of phytoplankton is a regular occurrence in Scandinavian forest lakes. These have been attributed to the very large temporal variations in the effective light climate; wide changes in water temperature, including winter ice cover; and nutrient availability (Ramberg, 1979; Arvola and Rask, 1984; Arvola, 1983). Despite regular monthly sampling over a two-year period, there appears to be no regular seasonal succession within the phytoplankton of Lake Chisholm, in keeping with most Tasmanian lakes (Tyler, 1974; King and Tyler, 1981b - but see also Cheng and Tyler, 1973a). Instead, the Lake Chisholm phytoplankton wax and wane capriciously with periodical virtual monospecific blooms. It is speculated that the differences between Lake Chisholm and its Scandinavian counterparts may lie in their contrasting thermal regimes. Lake Chisholm, at 42°S, 20° latitude closer to the equator than Finnish lakes, and exposed to a maritime rather than continental climate, is warm monomictic with tardy over-turn and no winter ice. In Finland, in comparison, there is rigorous dimixis. Even mid-winter in Tasmania sees sufficient day length, with a high enough solar elevation, for sufficient light for phytoplankton growth in Lake Chisholm. The lake's extended stratification means a relatively stable lake and uniform environment for many months of the year. The ecological pressures which favour phytoplankton succession in Finnish and Swedish lakes may therefore be considerably reduced in Lake Chisholm.

5.5 CONCLUSIONS

Lake Chisholm, Lake Barrington, the Pieman River reservoirs, and the lakes of the lower Gordon River, all display several features in common. All are dystrophic, with Lake Chisholm, Fidler, Morrison, and Sulphide Pool being polyhumic, and in all locations this produces shallow thermoclines with steep thermal gradients. The humus loading has other effects too, such as lowering pH, enhancing the solubility of reduced metal ions in the presence of sulphides and bicarbonate, and possibly adding to hypolimnetic anoxia through its decomposition.

The presence of meromictic or near-meromictic conditions within these waterbodies is largely a result of the long periods of thermal stratification, which creates a considerable stability against mixing. The morphometry of the lake basins and their degree of shelter from wind are also very important features which influence the extent of

wind-induced mixing in these lakes and reservoirs, and whether the buildup and maintenance of solutes in anoxic bottom waters, free from disruption by circulation, is possible. In those that are shallow compared with their surface area, such as Lakes Mackintosh and Morrison, incipient meromixis is quickly destroyed, as it is where exposure to prevailing winds occurs, such as in Lakes Rosebery and Pieman. However, where lakes are deep compared to their surface area, and sheltered by nearby hills and forests, such as Lakes Fidler, Barrington and Murchison, even weak chemical stratification can persist, or winter circulation will be sluggish, as in Lake Chisholm.

All the meromictic lakes and reservoirs had only small density differences due to solute concentration between their surface and bottom waters. A sufficient loading of decaying organics, either from pre-impoundment vegetation or from that produced within the waterbody; or of reduced iron and manganese, is also essential to create and later maintain an adequate density gradient to resist the effects of mixing, as in Lakes Murchison and Barrington, otherwise chemical stratification will break down, and holomixis eventuate. Likewise, the renewal of monimolimnetic salts by saline inflows into the lakes of the lower Gordon River is also required to maintain meromixis within them, and it is the inadequate supply of these since the river flow rates have been regulated that has caused its eventual demise. The original meromixis has now gone from both Lake Morrison and Sulphide Pool, and these should vacillate between holomixis and incipient meromixis in future, although changes in Sulphide Pool will be slower; while meromixis should remain in Lake Fidler for a long time yet, but the chemocline may sink too deep to support the array of micro-organisms which once straddled it. In Lake Chisholm, which is morphometrically predisposed to meromixis, it is only the lack of a solute-based density gradient in its bottom waters which mitigates against it becoming meromictic.

The study has shown three winter mixing or stratification patterns are possible in reservoirs. Firstly, chemical stratification, morphometry, and shelter between them may be insufficient to prevent full winter circulation, as occurs in Lakes Mackintosh and Rosebery. The anamolous stratification behaviour shown by Lake Gordon and Dartmouth Dam (Steane and Tyler, 1982; Welsh, 1984) provides a second pattern, where the morphometric restrictions imposed on mixing and on

the inflowing cold density currents allows for the undercutting of pre-existing bottom waters, but complete, wind-induced circulation is prevented. The third pattern, uninterupted chemical stratification all year, undisturbed by either winter circulation or density currents, is shown by Lakes Murchison and Barrington.

The dystrophic nature of these waters may favour a predominantly flagellate phytoplankton flora dominated by chrysophytes, as occurred in Lake Chisholm. This may result from their characteristic array of accessory photosynthetic pigments, an ecological tolerance to dystrophic waters, and to their motility. Blooms are frequent, but there is little evidence of structured seasonal successions of phytoplankton communities, possibly due to only limited environmental stresses occurring over time within the lake.

CHAPTER SIX

THE LIMNOLOGY OF DYSTROPHIC WATERS

6.1 CONCLUSIONS FROM THE STUDY

The study has demonstrated the strong influence dissolved humic substances have on the limnology of dystrophic waters, through their effects on the underwater light conditions, and their influence on thermal properties and water chemistry.

An overall perspective of the influence of gilvin on underwater light conditions can be gained from the investigations of standing freshwaters from Tasmania, north-east New South Wales; and south-east Queensland. In all three study areas it was the dominant attenuator of P.A.R., although tripton did contribute slightly in the waters of north-east New South Wales. Gilvin also causes the marked spectral modification of the incident light field, rapidly removing the shorter wavelength light. Even low gilvin concentrations, at barely detectable levels, have noticeable effects, absorbing blue light at depth, while its effects in polyhumic waters are severe. Likewise, gilvin dominates the optical properties of most freshwater coastal lagoon waters from the Bass Strait islands, and western and south-west Tasmania; as well as in Lake Chisholm, Lake Barrington, the reservoirs of the Pieman River (Chapter Three), and in the Gordon River lakes (King and Tyler, 1981b, 1982a, 1983; Croome and Tyler, 1984a, 1985a, Bowling and Tyler, 1986). Although turbidity may be an important attenuator in some parts of Australia (Section 2.5.5), gilvin is probably the major attenuator of light in most Australian inland waters.

The effect of high gilvin loadings on the thermal properties of lakes was shown by the coastal lagoons of the Strahan area of western Tasmania; by Lake Chisholm, the Pieman River reservoirs, and the Gordon River lakes. In the absence of wind induced mixing, strong thermal gradients quickly develop as heat is rapidly absorbed by their brown surface waters.

These gradients are of a transient nature only in the more exposed of the coastal lagoons, which would be classified by Lewis's (1983) system as discontinuous warm polymictic lakes. In comparison, Lake Garcia, along with Lake Chisholm, Lake Barrington (Tyler and Buckney, 1974) the Pieman River reservoirs, and the Gordon River lakes (King and Tyler, 1981b, 1982a, 1983) have thermal stratification of the warm monomictic type. Their dystrophic nature and shelter can result in extended periods of stratification and even reduced winter circulation.

Dissolved humic substances both directly and indirectly influence the chemistry of dystrophic waters. Hypolimnetic anoxia and sharp changes in redox chemistry with depth are partly consequent upon the long stable periods of thermal stratification, and possibly the microbial decomposition of some humus in the water (Salonen, 1981; Salonen *et al.*, 1983). However, oxygen never reaches 100% saturation in the humic waters of Tasmania. Because gilvin also increases the acidity of dystrophic waters, many dissolved chemicals are in a reduced state under these conditions, and exert an oxygen demand, even in epilimnetic waters (Patrick *et al.*, 1981). Humic substances probably enhance the reduction and solubility of iron and manganese in the chemically stratified reservoirs of this study, even in the presence of anions which would normally cause their precipitation. Additionally, chelation and subsequent precipitation may lower concentrations of calcium in some humic waters, such as Lake Chisholm and the Bass Strait island lagoons; while by decreasing pH, gilvin may lower the bicarbonate concentrations of dilute, unbuffered waters. Some chemical differences do occur between the coastal lagoon waters of the Bass Strait islands and those of western and south-west Tasmania (sufficient for Buckney and Tyler (1973a) to place them in different limnological categories), but this is due more to the higher salinities of the King and Flinders Island lagoons, rather than to the effects of their dissolved humic substances.

Although Finnish data suggest relationships between dissolved humic substances and the types and quantities of algae present in dystrophic freshwaters (Ilmavirta, 1980, 1982, 1983, 1984; Ilmavirta *et al.*, 1984; Arvola 1986), the single plankton net samples from north-east New South Wales, south-east Queensland, and western and south-west Tasmanian freshwaters are insufficient to show if similar relationships exist in these humic Australian lakes. However, the more extensive sampling undertaken on Lake Chisholm indicates that flagellates, and in particular chrysophytes may possibly dominate the phytoplankton communities of this lake. However

much more thorough metabolic investigations would be required to determine if any real differences do exist due to dystrophy.

The dystrophic nature of Lake Chisholm, the Pieman River reservoirs, and the backswamp lakes of the lower Gordon River is an important contributing factor in their tendencies towards meromixis, through the effects of humic substances on the thermal properties and chemistry of these waters. However other factors, such as shelter and basin morphometry, and the adequate resupply of solutes to the bottom waters, will eventually determine whether meromixis will be attained, maintained, or decline back to holomixis.

Thus, the study has shown the considerable effect dissolved humic substances have on many aspects of the limnology of dystrophic waters, and especially those of the western and coastal limnological provinces of Tasmania, where these predominate.

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APPENDIX ONE: PAPERS RESULTING FROM THIS THESIS

The following is a list of papers published or in press resulting from research undertaken as part of this thesis. Please note that papers 1) and 2) also contain research from my B.Sc.(Hons) thesis (Bowling, 1981). Photocopies of these papers are given in the following pages.

1) Bowling, L.C. and Tyler, P.A. (1984) Endangered lakes of scientific and cultural value in the World Heritage Area of South-west Tasmania. *Biol. Conserv.* 30, 201-209.

2) Bowling, L.C. and Tyler, P.A. (1986) The demise of meromixis in riverine lakes of the World Heritage wilderness of south-west Tasmania. *Arch. Hydrobiol.* 107, 53-73.

3) Bowling, L.C. and Tyler, P.A. (1984) Physicochemical differences between lagoons of King and Flinders Islands, Bass Strait. *Aust. J. Mar. Freshw. Res.* 35, 655-662.

4) Bowling, L.C., Steane, M.S. and Tyler, P.A. (1986) The spectral distribution and attenuation of underwater irradiance in Tasmanian inland waters. *Freshwat. Biol.* 16, 313-335.

5) Bowling, L.C. and Tyler, P.A. (1988) Lake Chisholm, a polyhumic forest lake in Tasmania. *Hydrobiologia.* (in press)

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